

**(REVISED)
FINAL**

**REMEDIAL INVESTIGATION
FEASIBILITY STUDY REPORT
FORMER MAINTENANCE AREA, PI-1
ZONE 1
FORT PICKETT, VIRGINIA**



Prepared for:
Department of the Army
U.S. Army Corps of Engineers
Norfolk District
803 Front Street
Norfolk, VA 23510-1096



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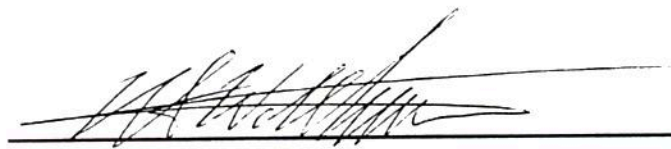
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Date

October 2001

EXECUTIVE SUMMARY

EA Engineering, Science, and Technology (EA) has prepared this Remedial Investigation/ Feasibility Study (RI/FS) report for the Norfolk District-U.S. Army Corps of Engineers (USACE) under USACE Contract No. DACA65-98-D-0027, Task Order No. 0003. This report addresses the Former Maintenance Area, PI-1, Zone 1 at Fort Pickett, Virginia. The objectives of the RI/FS process were to characterize the horizontal and vertical extent of constituents of potential concern (COPC) in affected media at PI-1, to assess potential risks to human health and the environment, and to evaluate the need for remedial action. This RI was completed in support of the Base Realignment and Closure Act (BRAC) program.

PI-1 encompassed about one acre and was formerly used as a vehicle maintenance area and service station, with no apparent underground tanks, from about 1937 to the 1960's. A concrete slab with soil-filled vehicle maintenance pit remains. An underground septic tank may remain onsite.

The field investigation included the completion of 8 soil borings, collection of soil samples for subsequent chemical analysis, surface soil sampling at 9 locations and subsequent analysis, installation of 2 monitoring wells, and collection of groundwater samples for chemical analysis. Analyses included EPA target compound list and target analyte list (TCL/TAL) organic and inorganic compounds, polycyclic aromatic hydrocarbons (PAHs), PCBs, total petroleum hydrocarbons (TPH), and dioxin/furans.

Human health COPCs included dioxin/furans, aluminum, and arsenic in surface soil, six metals in subsurface soil, and four pesticides and manganese in groundwater. The only case where non-carcinogenic risks were above the EPA cutoff of 1.0 was for child residents. The sole driver was manganese, which had a cumulative Hazard Index (HI) of 1.4 across soil and groundwater pathways. The target specific HI's for soil exposure did not exceed 1.0; therefore, potential non-carcinogenic effects from exposure to soil at the site are of no concern. Based on Fort Pickett background soil data, the manganese concentrations detected at the site in soil are consistent with background manganese concentrations. Therefore, no evidence exists to indicate a site-specific release of manganese in soil that could be considered responsible for manganese levels in groundwater. The occurrence of manganese in groundwater that result in a non-cancer risk greater than 1.0 is likely to be representative of background conditions considering groundwater quality in wells at other sites within Fort Pickett. Background groundwater quality data for Fort Pickett are not available at this time.

No cancer risks for any receptor exceeded EPA's target risk range of 10^{-6} to 10^{-4} . Ingestion of total soil and ingestion of groundwater for the resident (adult and child) and ingestion of surface soil for the adolescent trespasser were the only exposure scenarios for which potential cancer risks exceeded the de minimus 10^{-6} cancer risk. Arsenic and dioxin in soil, and aldrin, alpha-HCH, heptachlor, and heptachlor epoxide in groundwater were the primary risk drivers. Site arsenic concentrations were within the background data range for Fort Pickett. It is likely that the trace pesticides in groundwater are widespread in the area due to historical practices.

Several ecological COPCs had hazard quotient (HQ) values above one including aluminum, antimony, chromium, copper, lead, vanadium, zinc, and Total PAH based on the Tier 1 ecological risk assessment (ER). It was concluded that a Step 3 ERA should be completed. The Step 3 refined food web modeling was agreed to during a 13 June 2000 SMDP including EPA Region 3, the Ft. Pickett BRAC office, and the USACE.

Based on the Step 3 ERA, no observed adverse effect levels HQ values slightly exceeded 1.0 for aluminum, chromium, lead, vanadium and zinc for the shrew, rabbit, or robin. The concentrations of aluminum, chromium, and vanadium in surface soil were below or equal to background levels. This indicates the receptors of concern (ROC) could ingest soil at PI-1 and the risk would be at or below the risk from ingestion of background surface soil samples for these three metals. The majority of the lowest observed adverse effect level HQ values did not exceed 1.0. The small magnitude of no observed adverse effect level exceedances for lead and zinc suggest that risks to ecological receptors from exposure to COPC at PI-1 are negligible.

Based on the findings of the human health and ecological risk assessments, the existing constituent concentrations in soil and groundwater at the site do not pose a significant risk to human health and the environment. Therefore, following the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 as amended by the Superfund Amendments and Reauthorization Act of 1986 procedures, no alternatives were chosen for a Feasibility Study and based on this information, No Action is recommended at Site PI-1.

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LIST OF ACRONYMS AND ABBREVIATIONS

95UCLM	95th Percentile Upper Confidence Limit on the Mean
ADI	Average Daily Intake
AE	Average Exposure
AET	Apparent Effect Threshold
AOC	Area of Concern
ARAR	Applicable or Relevant and Appropriate Requirements
AT	Averaging Time
BCT	BRAC Cleanup Team
BRAC	Base Realignment and Closure
BTAG	Biological Technical Assistance Group
°C	Degrees Celsius
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COPC	Constituents of Potential Concern
DAF	Dilution-Attenuation Factor
DRO	Diesel Range Organics
EA	EA Engineering, Science, and Technology, Inc.
EBS	Environmental Baseline Survey
EPA	U.S. Environmental Protection Agency
EPC	Exposure Point Concentration
EPTC	s-ethyl dipropylthiolcarbamate
EQ	Ecological Quotient
ERI	Environmental Research, Inc.
°F	Degrees Fahrenheit
FS	Feasibility Study
ft	Foot/Feet
GRO	Gasoline Range Organics
HEAST	Health Effects Assessment Summary Tables
HHRA	Human Health Risk Assessment
HI	Hazard Index
HQ	Hazard Quotient
HSA	Hollow Stem Auger
ID	Inside diameter
in.	Inch(es)

LIST OF ACRONYMS AND ABBREVIATIONS (continued)

IP	Interface Probe
IRIS	Integrated Risk Information System
kg	Kilogram(s)
LADI	Lifetime Average Daily Intake
LOAEL	Lowest Observed Adverse Effect Level
LRA	Land Reuse Authority
m	Meter(s)
MCL	Maximum Contaminant Level
mg	Milligram(s)
MSL	Mean Sea Level
MS/MSD	Matrix Spike/Matrix Spike Duplicate
MSTI	Meridian Science and Technology, Inc.
NCEA	National Center for Environmental Assessment
NCP	National Contingency Plan for Oil and Hazardous Substances
NGVD	National Geodetic Vertical Datum
NOAEL	No Observed Adverse Effect Level
NOFA	No Further Action
NTU	Nephelometric Turbidity Units
OD	Outside Diameter
OSWER	Office of Solid Waste and Emergency Response
PAH	Polycyclic Aromatic Hydrocarbons
PA/SI	Preliminary Assessment/Site Investigation
PCB	Polychlorinated Biphenyls
PID	Photoionization Detector
POW	Prisoner of War
ppm	Parts per Million
PVC	Polyvinyl Chloride
QA/QC	Quality Assurance/Quality Control
RAGS	Risk Assessment Guidance for Superfund
RBC	Risk-Based Concentration
RCRA	Resource Conservation and Recovery
RfC	Reference Concentration
Rfd	Reference Dose
RI	Remedial Investigation

LIST OF ACRONYMS AND ABBREVIATIONS (continued)

RME	Reasonable Maximum Exposure
ROC	Receptors of Concern
RPD	Relative Percent Difference
SARA	Superfund Amendments and Reauthorization Act of 1986
SDWA	Safe Drinking Water Act
SF	Slope Factor
SIM	Selective Ion Monitoring
SMDP	Scientific/Management Decision Point
SQL	Sample Quantitation Level
SSL	Soil Screening Level
SVOC	Semivolatile Organic Compound(s)
TAL	Target Analyte List
TBC	To Be Considered
TCE	Trichloroethylene
TCL	Target Compound List
TCLP	Toxicity Characteristic Leaching Procedure
TEF	Toxicity Equivalent Factors
TEQ	Toxic Equivalency
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
TRV	Toxicity Reference Value
UCLM	Upper Confidence Level on the Mean
µg	Microgram(s)
USACE	U.S. Army Corps of Engineers
VAARNG	Virginia Army National Guard
VDEQ	Virginia Department of Environmental Quality
VOC	Volatile Organic Compound(s)
VPI	Virginia Polytechnic Institute

1. INTRODUCTION

1.1 PROJECT PURPOSE

EA Engineering, Science, and Technology (EA) has prepared this streamlined Remedial Investigation/Feasibility Study (RI/FS) report for the Norfolk District-U.S. Army Corps of Engineers (USACE) under USACE Contract No. DACA65-98-D-0027, Task Order Nos. 0003 and 0009. This RI/FS report addresses the Former Maintenance Area, PI-1, Zone 1 at Fort Pickett, Virginia. Figures 1-1, 1-2, 1-3, and 1-4 show in increasing detail the location and layout of the Former Maintenance Area.

The objectives of the RI process were to characterize the horizontal and vertical extent of constituents of potential concern (COPC) in affected media at PI-1 and to assess potential risks to human health and the environment. Data collected during the RI were used to support a Feasibility Study (FS), which evaluated the need for potential remedial alternatives.

Fort Pickett is undergoing a base closure process under the Base Realignment and Closure Act (BRAC) due to its selection by the BRAC 95 Commission. The strategy for investigation, remediation, and closure is overseen by the BRAC Cleanup Team (BCT), which includes the USACE-Norfolk District, U.S. Environmental Protection Agency (EPA) Region III, the Virginia Department of Environmental Quality (VDEQ), and the Fort Pickett Environmental Office. Site PI-1 was identified as a BRAC parcel for transfer or lease. The extent of this parcel is shown in Figure 1-3. This RI/FS was completed in support of the BRAC program.

1.2 OBJECTIVES OF RI REPORT

The objectives of the RI/FS report are to present a compilation and assessment of sufficient and appropriate data on the nature and extent of COPCs in soil and groundwater at the site, and information sufficient to support an informed risk management decision regarding the site and its future use. This effort included an initial evaluation of existing background information and the development of a conceptual site model (CSM) that characterizes contaminant pathways, receptors of concern (ROC), and data gaps.

Work completed on this project was conducted in accordance with guidance developed for the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA). CERCLA was implemented through the National Contingency Plan (NCP) for Oil and Hazardous Substances, as amended in 1990.

A previous version of the RI report was presented as *Remedial Investigation Report, Former Maintenance Area, PI-1, Zone 1 Fort Pickett, Final Revision*, June (EA 2000a). In addition, in response to BCT comments on this previous report and identified data gaps, a RI report addendum was prepared (EA 2000b). The report addendum addressed two EPA concerns on the human health risk assessment (HHRA) concerning inclusion of previous metal data collected by (WESTON 1998) and iron as a COPC. The addendum also included a Step 3 ecological risk assessment to address evidence of potential unacceptable risk to terrestrial receptors. The comments and responses on these reports are provided in Appendix L.

1.3 REPORT ORGANIZATION

Thus, this RI/FS report is a compilation of information presented in the previous version of the RI report (EA 2000a), the report addendum (EA 2000b), and information in response to subsequent BCT comments (Appendix L). This report is presented in 11 sections. Section 1 continues with a discussion on the background of the facility and site, and a review of the applicable or relevant and appropriate regulatory requirements (ARARs). The methodologies and objectives of the data collection activities, and documentation of the methods of data acquisition and quality control are presented in Section 2. Section 3 is a review of the physical setting of the project area. The presentations of a summary of the lab analytical results and assessment of the nature and extent of COPCs are included in Section 4. Section 5 is a review of the fate and transport of the COPCs in soil and water media. Section 6 evaluates the potential risks to human health from COPCs. Sections 7 and 8 evaluate potential risks to the environment from COPCs through a Tier I ecological risk assessment and a Step 3 refined food model, respectively. The feasibility study on evaluation of remedial action is discussed in Section 9. Section 10 presents a summary of the RI/FS and conclusions. References used in preparing this report are included before the appendices.

This report includes 12 appendices, named from A through L. Records of field activities and data collection are included in appendices A, B, and C. A summary of laboratory analytical data, summary reports for the validation of these data, and data quality analyses are presented in appendices D, E, and F. Appendices G and H present constituent toxicity information and calculations for the human health risk assessment. Likewise, backup information for the ecological risk assessment is provided in appendices I, J, and K. BCT comments on the previous versions of the RI report and responses to comments are presented in Appendix L.

1.4 PROJECT AREA DESCRIPTION¹

Fort Pickett is located in Brunswick, Dinwiddie, and Nottoway counties in southern Virginia (Figure 1-1 and 1-2). The installation was established by the U.S. Army in 1942 as a World War II training camp. Historically and through the present time, Fort Pickett has been primarily used to provide training facilities, maneuver training areas (including live fire and artillery ranges), base operations, and mobilization support for U.S. Army Reserve and U.S. Army National Guard units, as well as active component units of all services. It has also housed the Virginia Army National Guard (VAARNG), the Virginia Polytechnic Institute and State University Southern Piedmont Agricultural Experiment Station, and a variety of civilian and federal tenants.

Fort Pickett encompasses approximately 45,148 acres, of which 45,008 acres have been identified as BRAC property, subject to transfer or lease. The other 140 acres have been identified as a potential U.S. Army Reserve (USAR) enclave. Approximately 41,178 acres of the BRAC property will be transferred to the VAARNG. The remaining 3,829 acres of BRAC property, referred to as the Excess Property, remains the focus of continued environmental activities. This latter parcel is anticipated to be transferred for private use through the Fort Pickett Local Reuse Authority (LRA). In addition, approximately 11 acres associated with the wastewater treatment plant and the administrative and fueling facilities at the Allen C. Perkinson Municipal Airport were transferred to the town of Blackstone in 1996.

¹ The majority of text on the description of the project area and history was derived from the Fort Pickett Environmental Baseline Survey (EBS) report prepared by Woodward-Clyde (1997) and the Preliminary Assessment/Site Inspection (PA/SI) prepared by Roy F. Weston, Inc (WESTON 1998).

The project site, PI-1 (Former Maintenance Area), was formerly a service station and vehicle maintenance area. The site was identified as a BRAC parcel by Environmental Research, Inc. (ERI) (1997) due to potential environmental concerns. The area delineated for the BRAC property includes about two acres as shown in Figure 1-3. The study area for PI-1 encompasses about one acre and is located at the north end of Fort Pickett, adjacent to Business Route 460 (Figure 1-4).

Through a lease with the U.S. Army, the site (PI-1) and the surrounding property has been utilized by the Southern Piedmont Agricultural Research and Extension Center (SPAREC) since the mid 1970's. SPAREC utilized the property as a multi-purpose agricultural station focusing primarily on crop research. PI-1 is located on the northwest perimeter of the property, adjacent to fields where crop research is conducted. A concrete pad (all that remains of the old service station) has been used for hay storage.

Currently, all of the adjacent property surrounding PI-1 (1,187 acres) is in the process of being permanently transferred to the Virginia Department of Education and will continue to be utilized by SPAREC. Once PI-1 has been certified as environmentally suitable for transfer, it will be transferred to SPAREC and utilized to support their research mission.

PI-1 consists of 2.19 acres. Approximately half of the property has been utilized in the past to grow crops as part of the research activities. Also, there is a concrete pad (as mentioned above) that has been utilized for the storage of hay. The remaining areas of PI-1 are forested or are mowed occasionally; however, they are not utilized for research activities. Interviews with the SPAREC managers indicate that their plans are to continue using this property for the same purpose into the foreseeable future. There are no plans to utilize any of this property for residential purposes (Fort Pickett BRAC 2001).

1.5 INSTALLATION HISTORY

Prior to the American Colonial period, Southside Virginia and the area of Fort Pickett had been inhabited for more than 12,000 years by many Native American tribes of the Southeastern Woodland Indians. Exploration and settlement by the English began in the mid-1600s. By the mid-1700s, the area around the town of Blackstone had developed into a trading center. The American Revolution saw little military activity in the area. By the time of the American Civil

War, the Fort Pickett area had developed into a farming region for food and tobacco crops. There were many middle class and wealthy homes, and several churches, within the present installation boundary, but none of these remain. During the last year of the Civil War, units from both sides passed through the area, but no battles are known to have occurred within the Fort Pickett area. Little of major historical importance occurred in the area in the period between the end of the Civil War and the beginning of World War II (U.S. Army Garrison, Fort Pickett 1977). At the time of its acquisition by the War Department, the land consisted of subsistence-level farms mixed with pine and hardwood forests (The William and Mary Center et al. 1994). Fort Pickett was purchased by the U.S. government in 1941 and was activated as the Blackstone Military Area. In 1942, it was dedicated as Camp Pickett, named after General George E. Pickett who led the Confederate charge at Gettysburg. During World War II, Camp Pickett was a major combat training installation, with a peak number of 85,000 troops stationed at the installation during 1943. At that time, the camp had 1,600 buildings within the cantonment area and was served by 37 miles of surfaced roads, 125 miles of secondary roads, 11 miles of railroad, and four 5,000-ft-long airplane runways. Part of the installation was used as a prisoner-of-war (POW) camp for Axis soldiers captured in Europe (The William and Mary Center et al. 1994).

The U.S. Army discontinued combat training at the camp in the fall of 1944 and closed the facility at the end of World War II (Astore Architects and Urban Designers [Astore] 1991). It was briefly reopened during the period 1948 to 1949, closed again, and then reactivated in 1950 (the start of the Korean War) as a medical replacement U.S. Army training center. Armored unit training resumed during the early 1950s, the present training mission was assigned in 1960, semi-active status under Fort Lee was assigned in 1963, and the camp was redesignated as Fort Pickett in 1974 (U.S. Army Garrison 1977). From 1961 through the present, the active U.S. Army, USAR Command, VAARNG, U.S. Navy, and U.S. Air Force units have trained at the installation, with new construction and renovations continuing to support this training mission (Astore 1992).

Pesticides were applied to various areas of Fort Pickett from the early 1940's through closure in 1977. Pesticides were used to control insect hazards including ticks, mosquitoes, chiggers, and lice, and to control insects around buildings and grounds. Pesticides that were available for use during this period included the following: heptachlor epoxide, heptachlor, dieldrin, endrin, aldrin, alpha and gamma chlordane, 4,4-DDE, 4,4-DDD, and 4,4-DDT. The use of these pesticides was for the most part banned in the U.S. in the 1970's (USACE 2000).

1.6 PREVIOUS INVESTIGATIONS

1.6.1 Fort Pickett

As part of the BRAC process at Fort Pickett, Woodward-Clyde conducted an EBS of the base (Woodward-Clyde 1997). The EBS identified 311 BRAC parcels, which were defined as areas that can be segregated from surrounding areas based on environmental conditions or concerns. The EBS recommended that the base be divided into three geographic zones, each comprised of a number of BRAC parcels (sites) with specific environmental classifications. ERI conducted an Aerial Photographic Analysis of the base to identify BRAC parcels (ERI 1997). The photographic investigation identified 13 additional BRAC parcels, with assigned numbers in the format PI-##. The BCT conducted field surveys to identify areas of potential environmental concern based on visual information and known historic land use. The BCT identified another 15 BRAC parcels, with assigned numbers in the format BCT-##.

WESTON (1998) conducted a PA/SI of Areas of Concern (AOC) within Zone 1, including site PI-1. An AOC was defined as a BRAC parcel classified as Category 3 or higher. The purpose of the PA/SI was to determine if BRAC parcels can be closed under a "no further action" (NOFA) classification based on comparison of site sample data with human health and ecological risk-based criteria, or to identify if an RI/FS or removal action is warranted. The PA/SI included a site inspection, limited media sampling, and risk-based screening of analytical data.

1.6.2 PI-1

PI-1 (Former Maintenance Area) encompassed about one acre and is located at the north end of Fort Pickett, adjacent to Business Route 460 (Figure 1-4). PI-1 was formerly used as a vehicle maintenance area and service station. Vehicles, including buses, were frequently parked onsite for maintenance. ERI (1997) identified this site as a BRAC parcel in the aerial photographic survey. Photographs from 1949 and 1951 showed two buildings, several vehicles, and equipment at a probable maintenance area. As shown in Figure 1-4, areas with stains and disturbed ground were observed in the photographs in the southern portion of the site where vehicles, mechanical parts, and waste oil containers were possibly stored. The largest building present on the site was established in 1937. A 1955 aerial photograph showed the two buildings, but vehicles were not visible. Linear staining was visible south of the larger building. In a 1959

photograph, ground scarring was visible around the larger building. Staining of ground was visible near the south end. Notes on a 1964 photograph indicated that a maintenance building, which existed in 1963, was removed in 1964. Access roads led from the south end of the site to the former larger building location. PI-1 was identified by Woodward-Clyde (1997) as a Category 7 site in Zone 1 based on the site history, indicating that the property was not suitable for transfer until further site investigation was completed.

As part of the PA/SI, a site reconnaissance was conducted on 23 October 1997. The site was an approximately one-acre area with a fence located at the surrounding tree line. A building foundation pad was observed, but staining south of the pad was not visible. Stressed vegetation was observed on the west side of the larger building slab (WESTON 1998).

WESTON (1998) completed three soil borings at suspect locations around the larger building slab (Figure 1-4) with sampling intervals of 0-0.5 ft and 0.5-2 ft. Soil samples were analyzed for volatile and semivolatile organic compounds (VOC and SVOC) listed on the U.S. EPA target compound list (TCL), metals on the U.S. EPA target analyte list (TAL), and pesticides. A summary of detected analytes is provided in Table 1-1. Analytical results were compared against published, regional background levels, the U.S. EPA Region III Risk-Based Concentrations (RBC) for residential setting, EPA soil screening levels (SSLs) for soil ingestion (EPA 1996a), and ecological risk-based screening criteria.

The organic compounds di-n-butylphthalate, methylene chloride, and bis (2-ethylhexyl) phthalate were detected, but were attributed to laboratory contaminants. The pesticide alpha-chlordane was detected in one sample at 0.6 µg/kg. Trichloroethene (TCE) was detected in three samples at concentrations of 3-17 µg/kg. Seven polycyclic aromatic hydrocarbons (PAH) were detected in one sample at concentrations up to 0.7 µg/kg. These organic compound concentrations did not exceed human health or ecological screening criteria.

The concentrations of several metals in soil were above background levels, human health benchmarks, and/or ecological screening levels: aluminum, arsenic, beryllium, chromium, copper, iron, lead, mercury, nickel, selenium, vanadium, and zinc. WESTON (1998) recommended that a RI be conducted at site PI-1, including the analysis of TCL organics, TAL inorganics, and total petroleum hydrocarbons (TPH) in soil and groundwater.

1.7 POTENTIAL APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS (ARARS) AND TO BE CONSIDERED (TBC) CRITERIA

CERCLA guidance recommends that regulations, guidance, and other criteria that may relate to a CERCLA-style RI/FS be identified as early as possible in this process. The EPA has defined applicable requirements as “cleanup standards, standards of control, and other substantive environmental protection requirements, criteria, or limitations promulgated under Federal or State law that specifically address a hazardous substance, pollutant, contaminant, remedial action, location, or other circumstance at a CERCLA site.” The EPA has defined relevant and appropriate requirements as “substantive environmental protection requirements...promulgated under Federal or State law that, while not “applicable,”...address problems or situations sufficiently similar to those encountered at the CERCLA site that their use is well suited to the particular site.” The EPA has developed another category of requirements known as “to be considered” (TBCs), which includes nonpromulgated criteria, advisories, guidance, and proposed standards issued by Federal or State governments. TBCs are not potential ARARs because they are neither promulgated nor enforceable.”

This section summarizes regulatory issues that may apply to the project area. For this project, ARARs associated with regulatory issues must be considered on a site-specific, case-by-case basis. Specific regulations or criteria may be ARARs depending on future regulatory or corrective action implemented at the site. A potential list of ARARs is described below within three categories: chemical, location, and action specific.

1.7.1 Chemical-Specific Requirements

Chemical-specific requirements establish the health-based concentration limits in various environmental media for specific hazardous substances or pollutants. These requirements establish the protective levels for the constituents that may be incorporated into the remedial alternative evaluation when considering a specific remedial activity.

1.7.1.1 Federal and State Drinking Water Standards

Federal and state drinking water standards strictly apply only to the public water supply systems and not to groundwater supplies. Nevertheless, drinking water standards are commonly used as criteria for groundwater quality in RCRA/CERCLA cases.

The Federal Safe Drinking Water Act (SDWA), passed by Congress in 1974 and amended by the Safe Drinking Water Act Amendments of 1986 (PL 99-339), and subsequent revisions establishes National Primary Drinking Water Regulations that represent the maximum allowable level of selected contaminants that are economically and technologically feasible to achieve in water of public water systems. National Primary Drinking Water Regulations are developed by EPA based upon maximum contaminant level (MCL) goals. MCL goals are non-enforceable health goals of which there are no known or anticipated adverse effects on the health of humans utilizing the water source. The Safe Drinking Water Act tasks EPA with development of National Primary Drinking Water Regulations and MCL goals for drinking water contaminants. Pursuant to the Safe Drinking Water Act, EPA sets National Primary Drinking Water Regulations for drinking water contaminants as close to the MCL goals as is possible, based upon toxicological, economic, and engineering feasibility. EPA is also directed under the Safe Drinking Water Act to establish secondary National Primary Drinking Water Regulations, which address aesthetic considerations such as odor, turbidity, and taste.

In Virginia, the State has primacy for the Safe Drinking Water Act. The Virginia Department of Health (DOH) has adopted drinking water standards for public water supply systems. In this RI, MCLs in addition to EPA Region III RBCs (see Section 1.7.4.1 below) will be used to screen groundwater quality data to evaluate potential impacts to groundwater.

1.7.1.2 State Soil, Sediment, and Surface Water Quality Standards

Virginia has not presently established soil and sediment clean-up criteria, but develops site specific values on a case-by-case basis. State-developed values cannot in any case be less stringent than applicable federal guidelines. There are state and federal ambient surface water quality criteria; however, surface water quality criteria are not applicable for PI-1 considering that there are no surface water bodies or streams that cross or are near the site.

1.7.2 Location-Specific Requirements

Location-specific ARARs may affect or restrict remedial and project area activities. Generally, location-specific requirements serve to protect the individual site characteristics, resources, and specific environmental features.

1.7.2.1 Federal/State Environmental Resources

There are numerous state, federal, and local regulations that have been created for the protection and management of environmental resources. These regulations are inclusive of codes, laws, and standards designed to address development and other disturbance activities such as dredging, filling, and excavating in and around wetlands, floodplains, and adjacent lands. The project area is not known to include designated wetlands or floodplains. Additional associated compliance applies under other related regulations that become pertinent where there are mandated consistency and coordination reviews with other agencies including the Endangered Species Act of 1973 and State of Virginia endangered species regulations regarding federal and state listed threatened or endangered candidate species. These listings are handled by the Virginia Department of Conservation and Recreation, Natural Heritage Program; the U.S. Fish and Wildlife Service; and the National Marine Fisheries Service. Compliance with Section 106 of the National Historic Preservation Act includes consultation to determine if there are any potential impacts to properties listed in the National Register of Historic Places or Register-eligible sites as coordinated with the Virginia Department of Historic Resources.

1.7.3 Action-Specific Requirements

Although a remedial action plan has not been developed for PI-1, typically action-specific ARARs apply directly to remedial activities. These requirements pertain to storage, transportation, and disposal methods of hazardous substances according to RCRA requirements. These types of ARARs would apply only if remedial alternatives had been developed during the FS process. Several action-specific ARARs are discussed generically herein.

1.7.3.1 Resource Conservation and Recovery Act (RCRA)

RCRA and its implementing regulations provide technology-based requirements governing the identification, listing, storage, transportation, and disposal of hazardous wastes at active or potential hazardous waste facilities (generators, transporters, storage, or disposal facilities). RCRA requirements include groundwater protection, landfill permitting, design and performance standards, and standards for waste piles and surface impoundments. For this project, potential ARARs under RCRA relate to potential offsite disposal of waste materials from the project area. Specifically, 40 Code of Federal Regulations (CFR) 261 (Identification and Listing of Hazardous Waste), 40 CFR 263 (Standards Applicable to Transporters of Hazardous Waste), and 40 CFR 268 (Land Disposal Restrictions) will apply to removal and transportation of waste materials from the project area, which are identified as characteristic hazardous wastes based upon toxicity characteristic leaching procedure (TCLP) analyte concentrations (40 CFR 261). In addition, certain provisions of 40 CFR 264 (Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities), and 40 CFR 270 (EPA-Administered Permit Programs: The Hazardous Waste Permit Program) will apply to selected offsite hazardous waste disposal facilities. Offsite disposal of contaminated soil or water that exhibits RCRA hazardous characteristics must be at a Subtitle C RCRA-permitted disposal facility.

EPA Region III still retains authority for implementation of certain portions of the legislation, not covered by the State of Virginia, most notably many of the provisions of the 1985 Hazardous and Solid Waste Amendments, including the Corrective Action Program. The State of Virginia Department of Waste Management issues regulations under the Virginia Waste Management Act. Solid and hazardous wastes are defined in the Virginia Hazardous Waste Management Regulations (VR 672-10-1) and the Virginia Solid Waste Management Regulations (VR 672-20-10).

1.7.3.2 RCRA Land Disposal Regulations

The requirements of 40 CFR Part 268 (the Land Disposal Regulations) may be pertinent to the potential offsite disposal considerations for this project. The 8 November 1984 Hazardous and Solid Waste Amendments to RCRA serve as the basis for the Land Disposal Regulations. On 7 November 1986, the first of the Land Disposal Regulations went into place with the publication of treatment standards for spent solvent and dioxin-containing wastes. Shortly thereafter, EPA

issued regulations pertaining to land disposal of "California List" wastes. Regulations for the "first one-third" wastes were issued on 8 June 1989, and regulations for the "final one-third" wastes were proposed on 22 November 1989. The Land Disposal Regulations (often referred to as the "Land Ban") prohibit land disposal of regulated wastes that have been classified as RCRA listed hazardous wastes (in accordance with 40 CFR 261) or as RCRA characteristic wastes.

1.7.4 To Be Considered (TBC) Criteria

1.7.4.1 EPA Region III RBCs and SSLs

EPA Region III has defined RBCs (EPA 1999a) to screen sites during baseline risk assessment.² RBCs were developed using standard exposure scenarios with chemical concentrations corresponding to fixed levels of risk in drinking water and soil. Risk levels used were a Hazard Quotient (HQ) of 1, or a lifetime cancer risk of 1×10^{-6} , whichever occurs at a lower concentration. For noncarcinogenic chemicals, the RBC was divided by 10 to correspond to a HQ of 0.1, which ensured that chemicals with additive effects were not prematurely eliminated during screening. RBCs for soil in a residential setting will be used at PI-1 to screen soil analytical to evaluate potential impacts to soil at the site. RBCs for tap water will be used to evaluate potential impacts to groundwater. In the human health risk assessment, RBCs for soil in an industrial setting also were used to evaluate risk pathways for commercial and construction workers. In addition, the EPA Region III (2000a) soil screening values (SSLs) for migration to groundwater were used to evaluate potential contaminant leaching to groundwater. SSLs corresponding to a dilution-attenuation factor (DAF) of 20 were used.

1.7.4.2 Ecological Benchmarks

Ecological benchmarks for soil include values developed by Oak Ridge National Laboratory (Efroymson et al. 1997a, 1997b) and EPA Region III Biological Technical Assistance Group (BTAG) (EPA 1995a). Additional benchmarks were obtained from RIVM (1994, 1995). These benchmarks have not been promulgated as regulatory criteria, but are widely recognized for ecological screening levels at CERCLA sites. The minimum value of these benchmarks, which

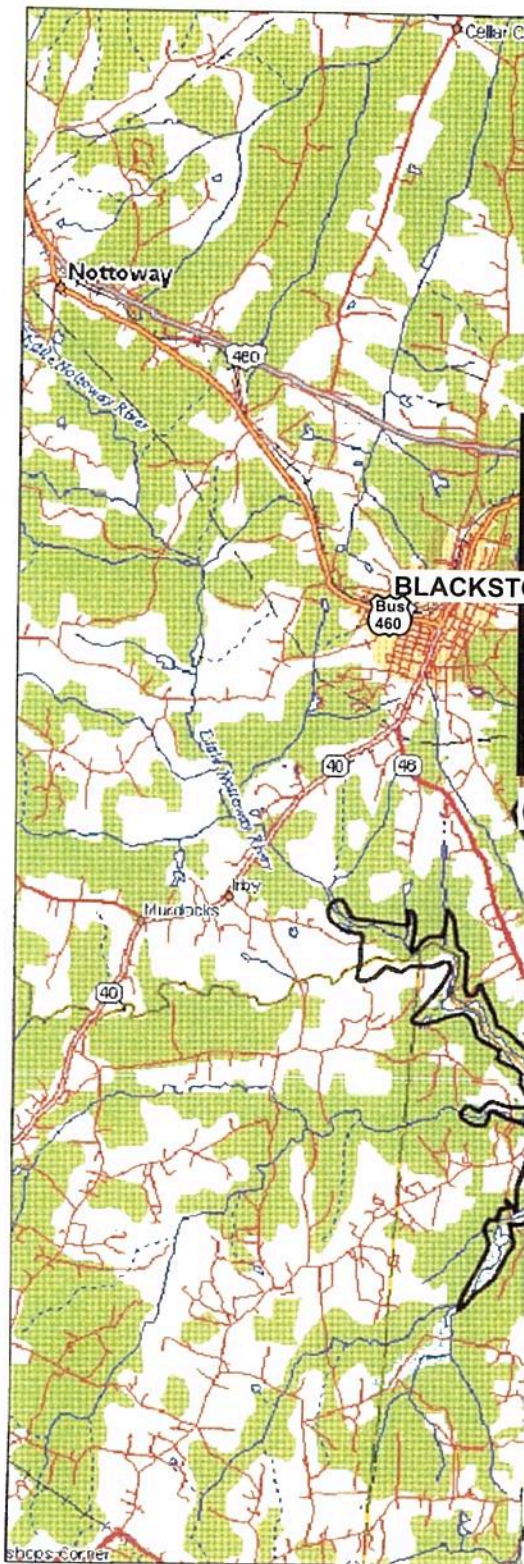
² Though more recent RBC values were available in EPA (2000a) released in April 2000, this RI report used RBCs from EPA (1999a), released in October 1999, due to the draft RI report completion time of March 2000. Slight differences in these RBC values would not have affected the data screening or risk assessment results.

was available for a particular analyte, was used to screen surface soil data to evaluate potential impacts to surface soil at the site.

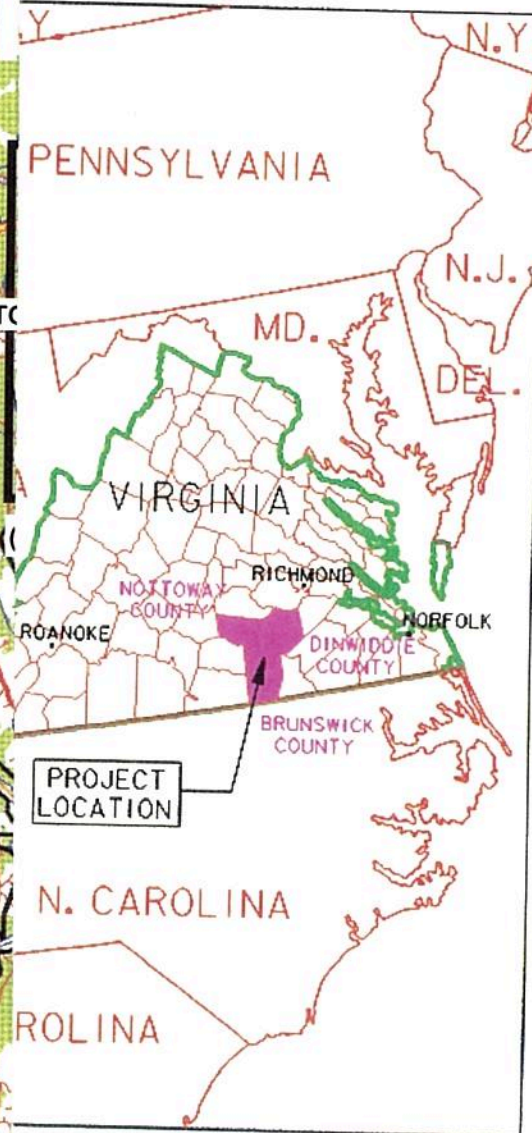
1.7.4.3 Background Soil Quality

The BCT developed statistically derived background soil levels for the post based on a background soil survey conducted at Fort Pickett in November 1999 (WESTON 1999a). This survey included 30 soil samples collected from eight locations across Fort Pickett that were not known to be impacted by current or historical U.S. Army or pre-Army agricultural activities. This survey established background soil concentration ranges for Fort Pickett for inorganics, PAHs, pesticides, and PCBs.

These background values were used for comparison against identified COPC concentrations in the risk assessment portion of the PI-1 RI. Table 1-2 presents the statistical summary of background total soil and surface soil results. For the human health risk assessment, Chapter 6, inorganic COPCs in total soil (i.e., surface and subsurface) were compared to the results of the background study including ranges of analyte concentrations, mean, and the 95 percent upper confidence limit on the mean (95 UCLM). In the environmental risk chapter, Chapter 7, the maximum concentrations of COPC in surface soil were compared to the mean and 95 UCLM of the surface soil background values as an evaluation of identified COPCs.

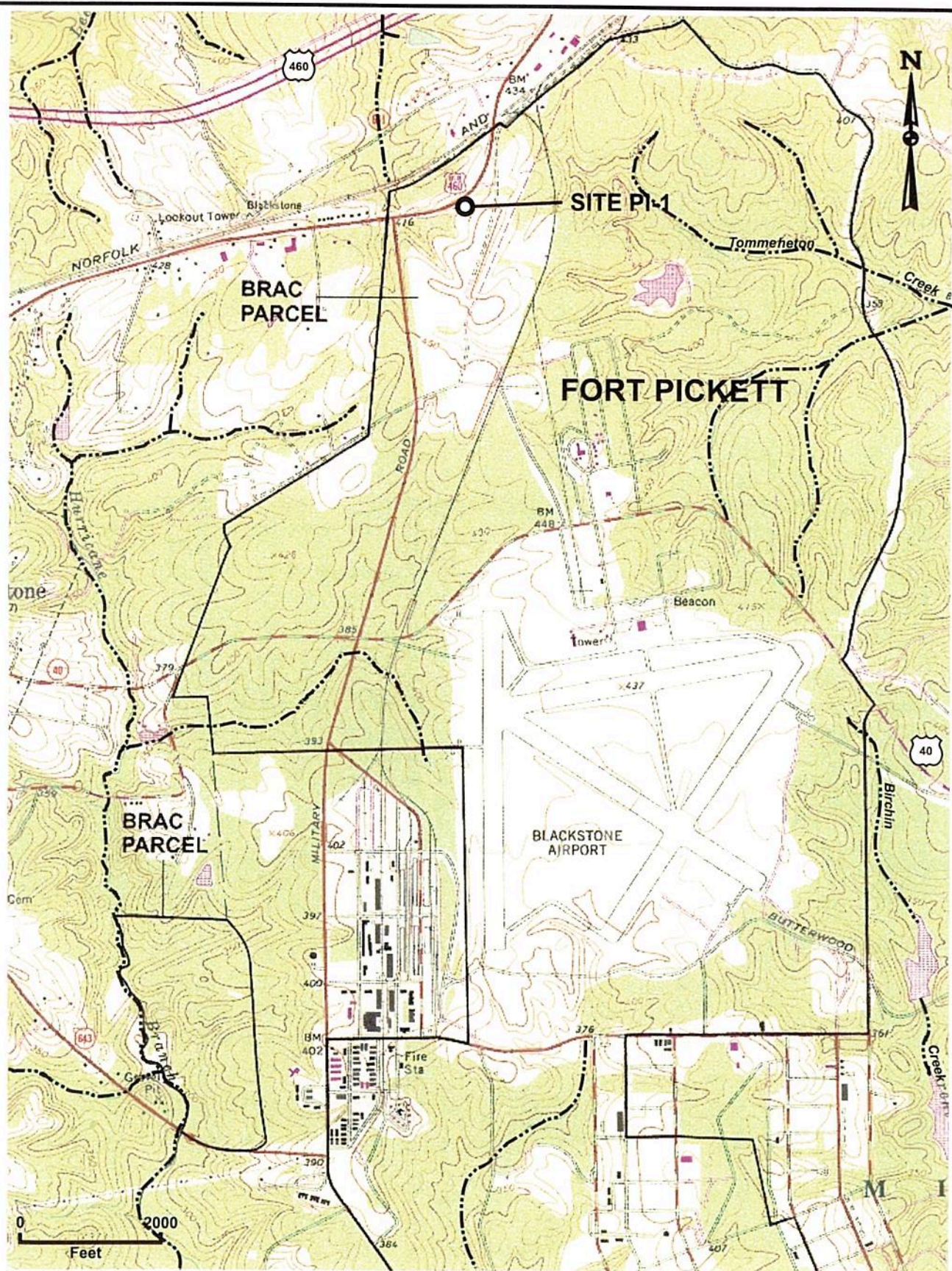


NOT TO SCALE



Source:
USACE - St. Louis District

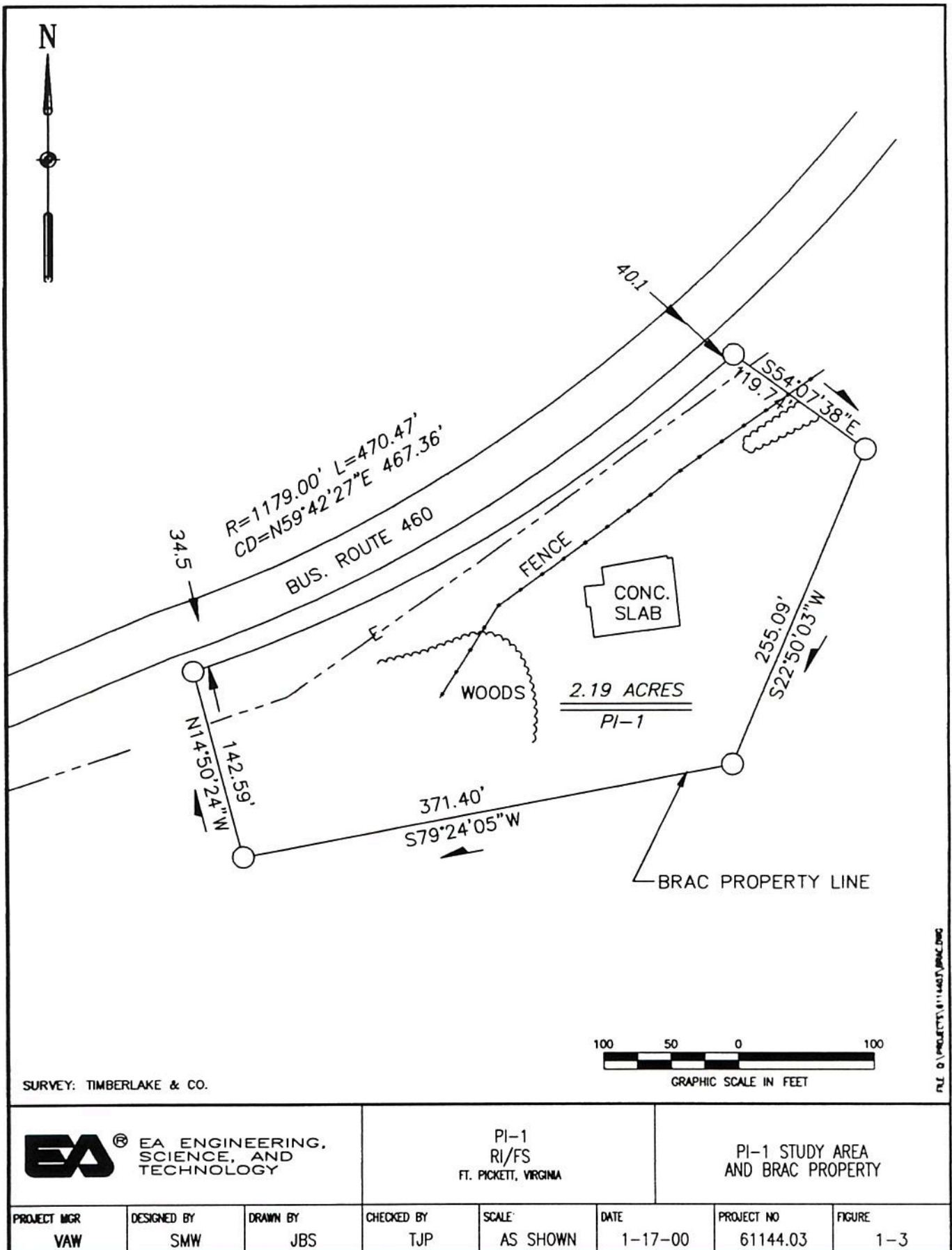




Base Map: USGS Blackstone East Quadrangle.

Figure 1-2. RI/FS Site Location, Fort Pickett, VA.



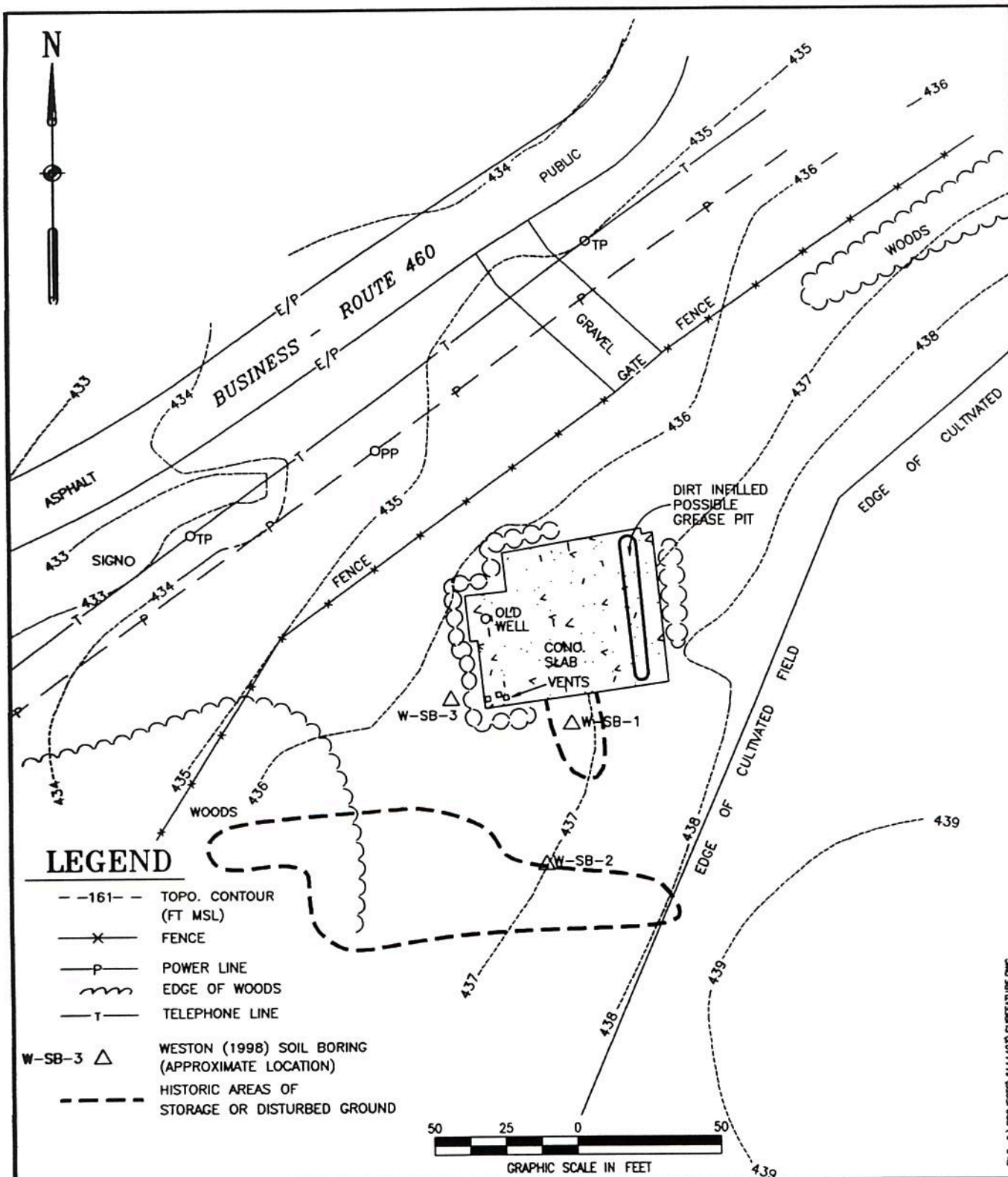


EA® EA ENGINEERING,
SCIENCE, AND
TECHNOLOGY

PI-1
RI/FS
FT. PICKETT, VIRGINIA

PI-1 STUDY AREA
AND BRAC PROPERTY

PROJECT MGR VAW	DESIGNED BY SMW	DRAWN BY JBS	CHECKED BY TJP	SCALE AS SHOWN	DATE 1-17-00	PROJECT NO 61144.03	FIGURE 1-3
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EA® EA ENGINEERING,
SCIENCE, AND
TECHNOLOGY

PI-1
RI/FS
FT. PICKETT, VIRGINIA

SURFACE FEATURES

PROJECT MCR
VAW

DESIGNED BY
SMW

DRAWN BY
JBS

CHECKED BY
TJP

SCALE
AS SHOWN

DATE
3-9-00

PROJECT NO
61144.03

FIGURE
1-4

TABLE 1-1 SUMMARY OF SOIL ANALYTICAL RESULTS AND COPCS FROM
WESTON (1998) BORING SAMPLES

Analyte	Sample Locations					
	PI1-SB1		PI1-SB2		PI1-SB3	
	0-6"	6-24"	0-6"	6-24"	0-6"	6-24"
TCL VOC (µg/kg)						
Methylene Chloride	23 B	27 B	24 B	16 B	14 B	74 B
TCE	<5	3 J	<6	<6	17	3 J
TCL SVOC (µg/kg)						
Di-n-Butylphthalate	<360	<410	120 J	150 J	<370	<430
Bis (2-Ethylhexyl) phthalate	<360	<410	<410	58 J	<370	<430
TCL Pesticides (µg/kg)						
α-chlordane	0.6 J	<2	<2	<2.2	<1.9	<2.1
PAH (µg/kg)						
Benzo(a)anthracene	<4	<4	<4	<4	0.4 J	<4
Benzo(a)pyrene	<4	<4	<4	<4	0.4 J	<4
Benzo(b)fluoranthene	<4	<4	<4	<4	0.7 J	<4
Chrysene	<4	<4	<4	<4	0.7 J	<4
Fluoranthene	<4	<4	<4	<4	0.4 J	<4
Phenanthrene	<4	<4	<4	<4	0.7 J	<4
Pyrene	<4	<4	<4	<4	0.4 J	<4
TAL Metals (mg/kg)						
Aluminum	16,600	58,200	41,800	45,000	7,890	37,900
Arsenic	2.6	5.9	5.3	5.4	35.1	4.7
Barium	55.4	53.7	52.1	31.5	278	27.2
Beryllium	<0.65	<0.73	<0.7	<0.75	1	0.79
Calcium	842	640	1,430	<250	7,200	934
Cadmium	<0.65	<0.73	<0.7	<0.75	<0.64	<0.75
Cobalt	1.8	2.8	2.6	1.6	3.8	1.6
Chromium	11.8	30.9	24.8	26.6	15.9	21.8
Copper	4.3	9.2	8.4	7.1	31.6	8.1
Iron	8,910	26,700	24,700	32,800	9,110	27,800
Potassium	540	1,020	791	735	448	1,050
Magnesium	636	957	924	676	438	920
Manganese	39.1	41.1	29.8	40.6	22	51.9
Mercury	<0.02	0.24	0.12	0.09	1.4	0.27
Sodium	<108	<122	<117	<125	241	<125
Nickel	4.5	12.5	8	7.2	10.5	7.2
Lead	27.8	70	42.9	47.6	22.6	32
Antimony	<1.1	<1.2	<1.2	<1.2	<1.1	<1.2
Selenium	<1.1	1.5	<1.2	<1.2	2	<1.2
Silver	<0.65	<0.73	<0.7	<0.75	<0.64	<0.75
Thallium	<2.2	<2.4	<2.3	<2.5	<2.1	<2.5
Vanadium	20.8	52.3	48.6	68.7	19.7	49.8
Zinc	15.2	25.4	21.2	23.2	20	62.7

BOLD TYPE: Analyte concentration was above human health screening value.

SHADED: Analyte concentration was above ecological screening value.

B: Analyte detected in blank.

J: Trace concentration below method detection limit.

TABLE 1-2 STATISTICAL SUMMARY OF BACKGROUND SOIL CONCENTRATIONS (a)

Chemical	Total Soil Data				Surface Soil Data			
	Min	Max	Arithmetic Mean	95% UCLM	Min	Max	Arithmetic Mean	95% UCLM
Organics (ug/kg)								
Acetone	6.0	97.0	19.4	28.7	6.0	97.0	21.6	89.6
Toluene	1.0	2.0	3.0	3.28	1.0	1.0	2.81	4.08
di-n-Butylphthalate	62.6	62.6	13.8	16.2	62.6	62.6	16.0	31.3
di-n-Octylphthalate	44.4	44.4	11.1	12.2	NR	NR	NR	NR
n-Nitrosodiphenylamine	53.1	18,100	820	722	292	824	146	9,470
Benzo(a)anthracene	4.5	4.5	9.51	10.2	4.5	4.5	8.76	11.0
Benzo(a)pyrene	5.3	5.3	9.54	10.1	5.3	5.3	8.86	10.5
Benzo(b)fluoranthene	4.5	4.5	9.51	10.2	4.5	4.5	8.76	11.0
Benzo(k)fluoranthene	4.9	4.9	9.53	10.1	4.9	4.9	8.81	10.6
Chrysene	4.2	4.2	9.50	10.2	4.2	4.2	8.72	11.2
Fluoranthene	3.8	3.8	9.48	10.3	3.8	3.8	8.67	11.4
Indeno(1,2,3-cd)pyrene	3.8	3.8	9.48	10.3	3.8	3.8	8.67	11.4
Metals (mg/kg)								
Aluminum	1,770	58,000	20,400	25,000	2,230	25,100	10,000	23,300
Arsenic	0.91	6.0	2.71	3.69	1.3	3.5	1.88	2.9
Barium	2.5	209	44.9	80.1	10.7	111	37.3	93.3
Beryllium	0.34	4.4	0.858	1.35	0.34	1.2	0.46	1.45
Calcium	55.1	499	145	248	55.1	499	146	322
Chromium	1.4	73.5	17.9	42.7	1.4	20.5	8.24	22.9
Cobalt	0.59	50.0	4.65	8.59	0.86	8.5	2.37	9.9
Copper	0.85	27.3	6.49	11.7	1.1	10.4	3.32	7.0
Iron	1,710	64,400	20,000	80,600	1,710	28,600	8,190	22,400
Lead	2.9	104.0	22.0	34.3	5.6	22.9	12.3	20.6
Magnesium	62	5,310	1,130	3,100	62	1,110	420	1,740
Manganese	2.5	1,270	217	735	23.4	325	98.3	411
Mercury	0.03	0.21	0.0374	0.05	NR	NR	NR	NR
Nickel	1.1	23.0	6.19	12.3	2.0	13.9	3.78	19.6
Potassium	158	6,460	1,220	2,900	158	766	315	930
Selenium	0.57	2.0	0.638	0.87	0.57	0.97	0.44	0.74
Vanadium	1.1	110.0	31.7	66.9	3.6	35.3	13.5	29.2
Zinc	20.7	95.1	26.1	67.3	25.0	43.8	11.6	49.3

(a) Background concentrations from WESTON (1999a). Non-detects were included at half the quantitation limit.

NR= No concentration reported.

2. FIELD METHODOLOGIES

2.1 INTRODUCTION

The field investigation for this RI at PI-1 commenced on 25 October 1999 and was completed on 12 November 1999. It included the completion of eight soil borings with collection of soil samples for subsequent chemical analysis, installation of two monitoring wells and collection of groundwater samples for chemical analysis, surface soil sampling, and a topographic survey. Sample locations are shown in Figure 2-1. Sampling and analysis were performed in accordance with the Final RI Work Plan (EA 1999), except as noted.

DataChem Laboratories performed the PAH analysis. Paradigm laboratories performed the dioxin/furan analysis. Severn Trent Laboratories performed the remaining sample analyses. Regional Probing Services, Inc. performed direct-push drilling. Bedford Well Drilling and Pump Co., Inc. installed monitoring wells. The site was surveyed by Charles H. Fleet & Associates, P.C.

2.2 SOIL SAMPLING

2.2.1 Rationale

Eight borings were completed for the collection of soil samples at locations where waste material may have been stored or disposed. One of the eight borings was completed as a monitoring well. In addition, one surface soil sample was collected within the swale located downslope of the site to evaluate surface runoff from the site. A summary of the soil sampling scheme is shown in Table 2-1, which indicates boring number, sample depth, analytes, and duplicate samples.

The sampling locations were based on historical photographs where areas of disturbed or stained ground, and former storage were indicated. Soil borings were located along the perimeter of the existing concrete pad, in the center of the concrete pad at a concrete hole (possible former hydraulic lift location), inside a dirt-filled former service or grease pit, downgradient west of the pad where a metallic anomaly was detected (possible septic drain field), and within a former storage area south of the pad. These sample locations, when combined with the former WESTON (1998) sample locations, provided adequate spatial coverage across the site.

The metallic anomaly was located by a metal detector survey of the unpaved areas outside the concrete pad. Over an area that extended approximately 50 ft around the perimeter of the slab,

the area was scanned with a Schonstedt Magnetic and Cable Locator (Model MAC-51B, Receiver frequency 40 Hz). Each side was walked along adjacent, parallel linear paths. One significant anomaly was detected west of the pad. This anomaly appeared to be approximately 3 ft long and 2-ft wide. The anomaly may be a septic tank as it is located downslope and in line with the toilet vent pipes on the slab. The associated boring SB-3 was completed adjacent to the anomaly to avoid hitting a possible hard object. The soil conditions in this boring appeared natural and not altered by the anomalous object. No other anomalies were detected around the site outside the concrete pad.

2.2.2 Sampling Equipment and Procedures

Subsurface soil samples were collected using direct-push or hollow-stem auger (HSA) drilling methods. Surface soil samples were collected at each boring location and also at a location in the drainage swale downslope of the slab. Surface samples were collected using a decontaminated stainless steel sampling trowel.

The direct-push method was performed by hydraulically pushing 4-ft long, 2-in. outside diameter (OD) stainless steel rods lined with 48-in. of Teflon-tubing. Subsurface samples were collected continuously. The soil boring that was converted to a well was completed by a mobile drill rig advancing 8-in.-OD HSA. Soil samples were collected continuously by driving 24-in. long, 2-in. OD split-barrel samplers. The sampler were attached to standard drilling rods and advanced up to 2 ft using a 130-lb hammer free-falling 30 in. Blow counts for each 6 inches of drive were recorded on the boring log. Soil borings were advanced to the water table interface at each sample location or until refusal on bedrock was encountered. Borings were completed at other well locations, but collected soil samples were not submitted for analysis. The handling of drill cuttings is described in Section 2.5.

Soil recovered from the sampler was initially screened for organic vapors in the field using a photoionization detector (PID). From each boring (not completed as a well) and MW-1 boring, three samples were collected for analysis. Soil samples were collected at the surface, as previously described, and at the water table interface. A third, mid-interval sample was collected based on elevated PID readings and/or visual evidence of suspect contamination (i.e., staining). The soil boring logs are presented in Appendix A.

The procedures for soil and quality assurance/quality control (QA/QC) sample collection were conducted in accordance with the Final RI Work Plan (EA 1999). Initially, the sample container for VOC analysis was filled. The remaining soil was homogenized in a stainless steel bowl prior

to filling the appropriate sample jars for other analysis. Samples for QA/QC included duplicates, rinsates, a field blank, trip blanks, and matrix spike/matrix spike duplicates (MS/MSD). QC samples were filled from the same discrete sample or composite mixture as the field samples. In addition, QA splits of five percent of the samples were submitted to DataChem Laboratories, which was contracted by USACE (Table 2-1).

Samples were placed in clean, laboratory jars and were shipped overnight to the appropriate laboratory for analysis. The samples were analyzed for compounds listed on the EPA TCL and TAL: VOCs (Method 8260), SVOCs (Method 8270), pesticides (Method 8081), polychlorinated biphenyls (PCBs) (Method 8082), PAHs (Method 8270 using selective ion monitoring [SIM]), metals (SW846), and cyanide (SW846). Samples were analyzed also for TPH including diesel range and gasoline range organics (DRO and GRO) (Method 8015M). In addition, four surface soil samples collected near the pad and former storage area were analyzed for dioxin/furan isomers (Method 8290). One surface and one subsurface soil sample from a boring were analyzed for total organic carbon (TOC) (Method 9060) and grain size distribution (ASTM method D 422). TOC and grain size provided a general characterization of soil for the site. These parameters provide information for fate and transport analysis, and for bioavailability of contaminants in soil. The subsurface sample was collected from the saturated groundwater zone.

2.3 GROUNDWATER

2.3.1 Rationale

Based on the geologic setting, groundwater was expected to occur in the overburden above bedrock and the water table was expected to be about 20 ft deep. Four monitoring wells were proposed. The wells were to be installed to a depth of 10 ft below the encountered water table.

Two monitoring wells (MW-1 and MW-2) were installed downgradient of the concrete pad and a former storage area; however, bedrock (i.e., HSA refusal) was encountered before reaching the water table at two upgradient locations (MW-3 and MW-4). At the time of this RI fieldwork, the two downgradient wells were considered sufficient to evaluate potential onsite impacts to groundwater. USACE was informed of this situation. The installation of upgradient wells within bedrock may be considered later based on the results of this study. A summary of the well sampling scheme is shown in Table 2-2, which indicates analytes and duplicate samples.

A third, onsite well was available for water level gauging and measurement of the groundwater gradient. The former water supply well was about 25 ft deep, 2.5 ft in diameter, and constructed

of cylindrical tiles. This well was probably installed when the site building was constructed in the 1930's.

2.3.2 Monitoring Well Installation

Borings for well installation were completed using an Ingersoll-Rand A300 drill rig to turn 8-in. OD HSA as described above. Soil samples were collected for lithology characterization and PID screening. Well completion diagrams are presented in Appendix A. The wells were constructed by the following procedures:

- Riser and screen consisted of 2-in. inside diameter (ID), Schedule 40 polyvinyl chloride (PVC) well casing and 0.020-in., machine-slotted PVC well screen.
- Connections were flush jointed and threaded, and each well had 15 ft of 0.020-in. slot well screen with an endcap.
- The screen interval was placed at the bottom of the wells such that at least 5 ft of screen was above the apparent water table.
- An expandable well plug was placed at the top of the riser.
- Well casing was extended about 2 ft above the ground surface.
- Clean No. 2 Morie sand was used to construct the filter pack from the bottom of the well to 3 ft above the top of the screened interval.
- One foot of finer sand was placed on top of the filter pack to prevent migration of bentonite into the filter pack.
- A 3-ft-thick bentonite seal was installed in 12-in. lifts after the filter pack. Bentonite seals were composed of commercially available 100 percent coarse grade bentonite.
- Following placement of each bentonite lift and before additional materials were placed, clean, potable water (from approved water supply) was poured down the annular space to hydrate the bentonite for 30 minutes.
- A 1-ft layer of fine sand was placed above the bentonite to act as a barrier to downward grout migration.
- Auger flights were progressively removed from the borehole allowing the auger to act as a tremie pipe. The depth to the top of the filter pack or bentonite was measured with a weighted measuring tape.
- The annular grout seal was placed above the fine sand barrier. The grout consisted of cement-bentonite grout, composed by weight of 10 parts cement (Portland cement, type I) to ½ part bentonite with a maximum of 7 gal of approved potable water per 94-lb bag of

cement. Bentonite was added after the required amount of cement was mixed with water. Additives or borehole cuttings were not mixed with the grout.

- The above-grade PVC riser was protected with a steel, outer casing (6-in. by 5-ft long) cemented in place about 2.5 ft above ground surface immediately following grouting.
- The protective casing was surrounded with a 1-ft square concrete pad about 6 in. thick. The protective casing has a hinged cover that is secured with a padlock.
- The casing was painted orange and four 3-in. diameter steel posts were placed radially around and 3 ft from the casing.
- Each post was 5-6 ft long and was placed 2-3 ft below the ground surface and extended 3 ft above the surface. The posts were painted orange.
- Corrosion-resistant tags with stamped well identification numbers were affixed to the well covers.

2.3.3 Well Development

Proper well development is necessary to remove silt from the well and to ensure that representative groundwater samples will be obtained from the well. The new wells were developed within one week after each well was constructed, but no sooner than 48 hours later. Development water was handled in accordance with procedures described in Section 2.5.

The depth to static water level and total well depth were measured from the top of casing prior to development at each well. Initially, the wells were mechanically surged with a submersible pump, but the wells soon ran dry. The low-yielding wells were hand bailed.

The wells were continuously hand bailed and water quality parameters were measured using a YSI 610D meter for every well volume removed. The wells were hand bailed dry several times. The bailer was moved through the water column to surge the well. Stabilization of water quality parameters over three consecutive readings was used to ascertain if well development was complete: ± 0.1 pH units, temperature $\pm 1^{\circ}\text{C}$, conductivity ± 5 percent, and turbidity less than 10 nephelometric turbidity units (NTU). Satisfactory well development was achieved and noted on the Field Records of Well Development, which are presented in Appendix B.

Physical characteristics such as odor, color, turbidity, and the presence of separate phase liquids were noted throughout development operations. At the end of development, a 1-pint jar sample of well water was collected from the pump discharge. The jar was labeled with the well number

and date. The jar was shaken and then photographed with back light to show water clarity. Photographs are provided in Appendix B.

2.3.4 Well Purging and Sampling

Groundwater samples were collected from wells two weeks after well development. Well purging and sampling methods followed low-flow sampling procedures defined by U.S. EPA (EPA, 1996b). Initially, the well cap was removed and a headspace reading was taken with a PID to assess the presence of organic vapor in the well. PID readings were recorded on sampling records. The water level in the well was gauged using a decontaminated interface probe (IFP). No separate phase liquids were encountered during gauging of the wells. Purge water was managed in accordance with procedures described in Section 2.5.

Purging of wells is necessary to remove groundwater in the well that has been isolated for an extended period of time and may no longer be representative of the aquifer. A Redi-Flow[®] submersible pump with check valve and dedicated Teflon FEP-lined tubing was used to purge the monitoring well prior to sampling. The well was pumped at a rate such that drawdown did not exceed 0.3 ft (e.g., start at 0.5 L/min and adjust so as not to exceed 0.3 ft. drawdown). Temperature, pH, conductivity, dissolved oxygen, and oxidation-reduction potential were monitored using a YSI 610D meter and flow-through cell during pumping at a rate of one reading every 5 minutes. Turbidity was monitored using a Hach turbidity meter. An in-line tee diverted groundwater for turbidity measurements. Pumping continued until these parameters stabilized over three consecutive readings: pH ± 0.1 units, temperature $\pm 1^{\circ}\text{C}$, conductivity ± 3 percent, dissolved oxygen ± 10 percent, redox potential ± 10 mV, and turbidity ± 10 percent. Field Records of Well Purging and Sampling are presented in Appendix B.

At the end of the purging procedure, the appropriate laboratory bottles were filled. The samples were analyzed for TCL and TAL compounds: VOC (Method 8260), SVOC (Method 8270), pesticides (Method 8081), PCBs (Method 8082), PAH (Method 8270 using SIM), total metals (SW846), and cyanide (SW846). Samples were analyzed also for TPH (DRO and GRO). Samples from MW-2 were collected for dissolved (filtered aliquot) and total TAL metals because turbidity was more than 15 NTU. Samples for QA/QC included a duplicate, rinsates, a field blank, trip blanks, MS/MSD, and a split QA sample submitted to DataChem Laboratories (Table 2-2).

2.3.5 *In Situ* Permeability Testing

In situ permeability testing (slug testing) was performed on the two wells after well development and sampling, and return of the water level to static conditions. Slug tests were conducted to estimate *in situ* hydraulic conductivity of the screened interval at each well location. Slug tests, consisted of both falling and rising head tests.

For slug testing, a clean pressure transducer was placed in the well and calibrated according to the manufacturers specifications. The water level in the monitoring well was measured with a water level indicator prior to the initial insertion of the slug. A clean slug (1 in. x 36 in.) was inserted into the selected well, which consisted of a length of PVC pipe filled with sand, and sealed on both ends. In performing a falling head test, the slug was quickly submerged into the

well, when an elapsed time count began. Water levels were automatically monitored by a computerized *In Situ* Hermit data logger until the level recovered to not less than 80 percent of the original static level or 60 minutes had elapsed. After the water level re-equilibrated, a rising head test was performed by quickly removing the slug from the well and automatic monitoring was repeated.

The data obtained from performing the rising head and falling head slug tests were downloaded from the data logger to a personal computer for further analysis with AQTESOLV®. The Bouwer and Rice method option of AQTESOLV® was used to determine hydraulic conductivity from these data. Slug test data analyses are presented in Appendix C.

2.4 DECONTAMINATION PROCEDURES

2.4.1 Soil Boring Rig and Downhole Tools

Equipment used during HSA drilling was steam cleaned prior to use. Equipment (i.e., split-barrel samplers or soil cores) was kept off of the ground and placed on clean plastic sheeting. Equipment that was decontaminated included augers and samplers. Following a drilling operation and before moving to the next drilling location, augers and downhole sampling equipment were steam cleaned at the designated decontamination area.

2.4.2 Sampling Equipment

Soil sampling equipment (i.e., spoons, triers, trowels, bowls, etc.) was cleaned prior to use in the field. Because the direct-push samplers had Teflon liners, the decontamination of the sampling spoons between samples was simplified and included a detergent wash, tap water rinse, and a deionized water rinse. Wherever possible, sampling equipment was dedicated to a single location to minimize potential for cross-contamination. Non-dedicated sampling equipment was decontaminated as described below.

Stainless Steel: This included spoons, spatulas, trowels and bowls, hand augers, split-barrel samplers, and other stainless steel equipment used for field activities.

1. Wash thoroughly using a brush and laboratory-grade glassware detergent (e.g., Alconox), plus tap water to remove particulate matter and surface films. Steam cleaning was conducted on spoons, and augers.
2. Rinse thoroughly with tap water from the designated decontamination water supply source.
3. Rinse with 10 percent nitric acid diluted with deionized water for samples analyzed for metals.
4. Rinse with tap water.
5. Rinse twice with pesticide grade methanol by spray bottle and allow to air dry. Other solvent rinses such as hexane or isopropyl alcohol were not used. Glass spray bottles were not available during fieldwork.
6. Triple rinse with deionized water.
7. Air dry.
8. Wrap equipment in aluminum foil (shiny side out) and store in plastic garbage bags to prevent contamination during storage and/or transport to the field.

2.5 INVESTIGATIVE-DERIVED WASTES

Drill Cuttings and Well Water

Waste material was screened using a PID. The following criteria were utilized:

- Drill Cuttings: A clean glass jar was filled three-fourths full of the waste soil, capped with aluminum foil and a lid, and allowed to equilibrate for 15 minutes in a warm setting. A headspace reading was then taken through the foil. PID readings

were less than 5 ppm above ambient background. The soil from well holes was stockpiled onsite near the area of the well and covered with plastic, and borings were backfilled.

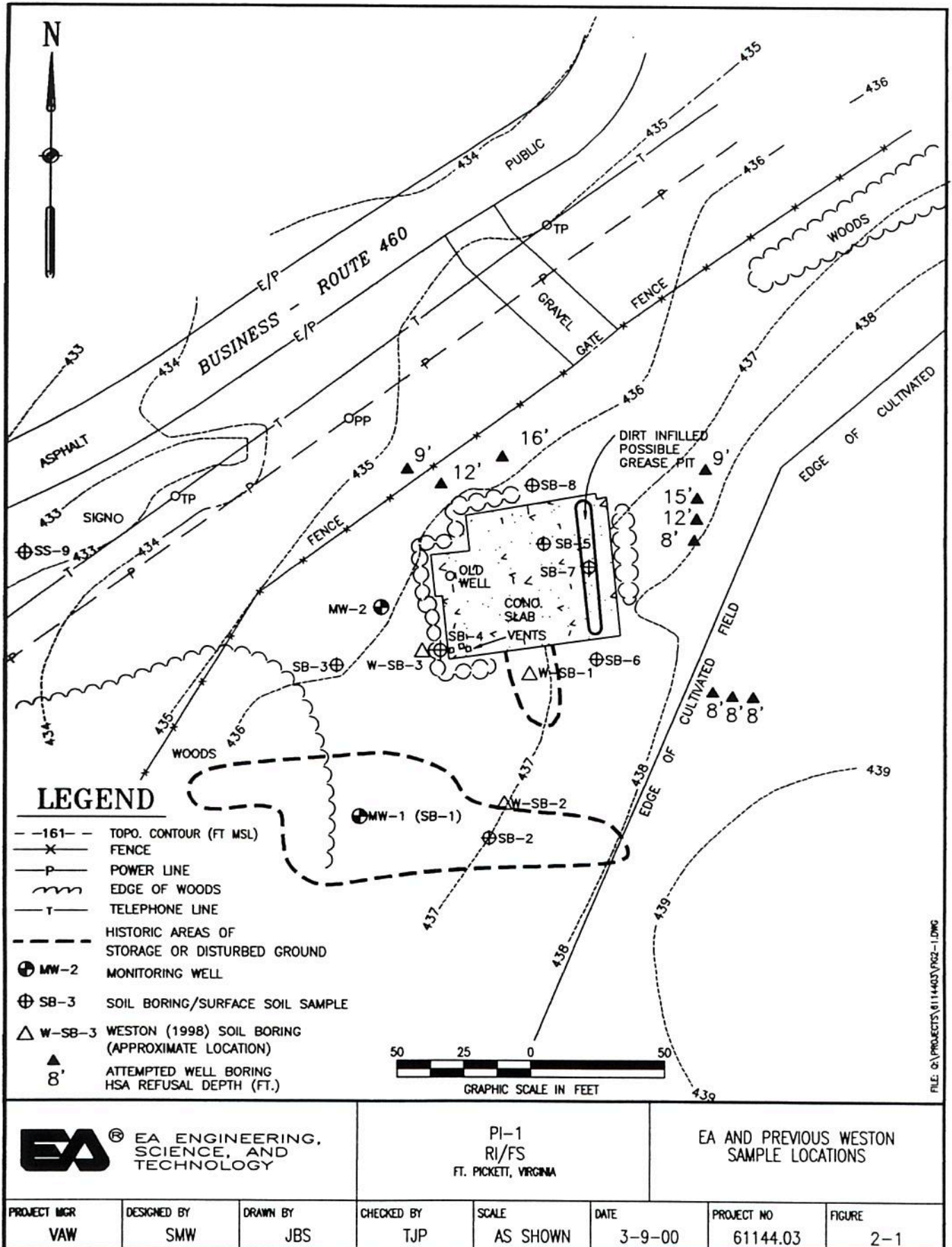
- **Decontamination and Well Water:** A clean glass jar was filled three-fourths full with the waste water in question, capped with aluminum foil and a lid, and allowed to equilibrate in a warm setting for 15 minutes. A head space reading was taken through the foil. If the reading was less than 5 ppm above ambient background (as measured onsite), the water was disposed on the ground at the site, downgradient from sampling activities, in a manner such that it infiltrated into the ground. Decontamination and well water generated at Site PI-1 did not exhibit elevated PID headspace readings, therefore, this water was disposed onsite.

Other Waste

Other wastes generated during decontamination activities, including discarded personal protective equipment, aluminum foil, and other debris, were collected and containerized in double plastic bags. These solid waste materials were presumed to be non- RCRA hazardous and, therefore, were disposed of in a solid waste container/dumpster for proper disposal.

2.6 SURVEY

The project site was surveyed to define the topography and major land surface features. In addition, the locations of soil borings, wells, and surface samples were surveyed. The survey data were used to establish horizontal and vertical control of monitoring well and boring locations. Elevations were surveyed with respect to National Geodetic Vertical Datum (NGVD88). Horizontal locations were surveyed with respect to 1983 VA State Planar Coordinates. The location surveying conformed to standard third order accuracy for wells, which is consistent with other environmental investigations (i.e., remedial investigations). A topographic map of the site was developed with a scale of 1 in. equals 40 ft and a 1-ft contour interval.



FILE: Q:\PROJECTS\6114403\FIG2-1.DWG

TABLE 2-1 SUMMARY OF SOIL SAMPLING SCHEME, PI-1, FORT PICKETT

Boring	Sample Depth (ft)	Field Samples							QA/QC Samples		
		TOC (9060), grain size	Dioxin/furans (8290)	TCL VOC (8260)	TCL Pesticides/PCBs (8081, 8082)	TCL Non-PAH SVOC (8270)	PAH by SIM (8270)	TPH (GRO/DRO) (8015M)	TAL metals, cyanide (6010/7000/9012)	Duplicate	QA Split
SB-1	0.5		*	*	*	*	*	*	*		
	12			*	*	*	*	*	*		
	20			*	*	*	*	*	*	*	
	26	*									
SB-2	0.5		*	*	*	*	*	*	*		
	6			*	*	*	*	*	*		
	11			*	*	*	*	*	*		
SB-3	0.5		*	*	*	*	*	*	*		
	10			*	*	*	*	*	*		
	18			*	*	*	*	*	*		
SB-4	0.5			*	*	*	*	*	*		
	10			*	*	*	*	*	*		
	17			*	*	*	*	*	*		
SB-5	0.5			*	*	*	*	*	*		
	4			*	*	*	*	*	*		
	15.5			*	*	*	*	*	*		*
SB-6	0.5		*	*	*	*	*	*	*	*	*
	6			*	*	*	*	*	*		
	11.5			*	*	*	*	*	*		
SB-7	0.5			*	*	*	*	*	*		
	6			*	*	*	*	*	*		
	14			*	*	*	*	*	*		
SB-8	0.5	*		*	*	*	*	*	*		
	8			*	*	*	*	*	*		
	14			*	*	*	*	*	*		
SB-9	0.5			*	*	*	*	*	*		

TABLE 2-2 SUMMARY OF GROUNDWATER SAMPLING SCHEME, PI-1, FORT PICKETT

Well Name	Well Depth (ft)	Field Samples						QA/QC Samples	
		TCL VOC (8260)	TCL Pesticides/PCBs (8081, 8082)	TCL Non-PAH SVOC (8270)	PAH by SIM (8270)	TPH (GRO/DRO) (8015M)	TAL total metals, cyanide (6010/7000/9012)	TAL dissolved metals (6010/7000)	QA Split
MW-1	26	*	*	*	*	*	*	*	*
MW-2	24.5	*	*	*	*	*	*	*	

3. PHYSICAL CHARACTERISTICS OF THE STUDY AREA³

3.1 PROJECT AREA SETTING

3.1.1 Location

Fort Pickett lies in the southern Piedmont region of Virginia, and its area is divided among Brunswick, Dinwiddie, and Nottoway counties. The installation is located approximately 60 miles southwest of Richmond and 35 miles west of Petersburg on U.S. Route 460. The region surrounding the installation is agricultural, and the closest town is Blackstone, whose center is approximately 1 mile west of the installation boundary, near the northwest part of the installation. Site PI-1 is located on the northern border of the post (Figure 1-2).

3.1.2 Demographics

At the time of the 1990 census, the combined population of Brunswick, Dinwiddie, and Nottoway counties was 51,940. Populations in all three counties have been declining overall since the 1960 census, with the overall decrease for this period being 5.7 percent. Blackstone is the only adjacent town. It is located approximately 1 mile west of the installation. The population of Blackstone was 3,497 at the time of the 1990 census, a decrease of 4.4 percent from 1960.

Tobacco, cash grain crop, livestock farming, and timber harvesting are important agricultural commodities to the region and will probably remain so into the future. Small industry is a subordinate part of the economy but has been increasing. A lumber treatment plant and textile, apparel, and furniture-making facilities are situated in or near Blackstone (Astore 1991).

3.1.3 Topography and Drainage

Fort Pickett is located within the Piedmont Province, which is both a physiographic and geologic designation. This province trends through central Virginia from north to south, and is bounded by the Blue Ridge Province on the west and the Coastal Plain Province on the east (Powell and Abe 1985). Elevations on the installation range from approximately 190 ft above mean sea level

³ Project setting information was derived from the Fort Pickett Environmental Baseline Survey report prepared by Woodward-Clyde (1997) and the PA/SI prepared by WESTON (1998).

(MSL) along the Nottoway River at the southeastern installation boundary to 450 ft above MSL at the Blackstone Army Airfield. Most of the topography on the installation is gently rolling, with steeper slopes and flat, marshy floodplains along the principal drainages (Astore 1991).

Most of the installation, except for its northeast corner, is drained by the east-flowing Nottoway River and three of its first-order tributaries: Tommeheton Creek, Birch Creek, and Hurricane Branch (Figures 1-1 and 1-2). These tributaries are fed by a dendritic system of perennial and intermittent streams. Butterwood Creek and its tributaries drain the northeast corner of the installation. The overall direction of drainage is to the east. The Nottoway Reservoir (approximately 390 acres), in the southwest corner of the installation, is the largest surface water body. It is formed by damming the Nottoway River at Route 460 and serves as the water supply for the installation and the town of Blackstone. Smaller bodies of water on the installation are Twin Lakes on Butterwood Creek, Birch Lake on Birch Creek, and Tommeheton Lake on Tommeheton Creek.

As shown in Figure 3-1, the ground elevation of the site is about 437 ft MSL. The ground surface slopes down at an approximate 2.5 percent grade to the northwest. Runoff is directed northwest towards a shallow swale that parallels Business Route 460.

3.1.4 Climate

The climate of the area is temperate. The normal prevailing wind direction is westerly, with occasional periods of winds from the southeast and east. Moderate temperatures are the norm, with relatively mild winters and warm summers with occasional hot days. The average summer temperature is 76.6 degrees Fahrenheit (°F) and the average winter temperature is 38.7°F. Precipitation is fairly evenly distributed throughout the year, but short dry periods occur in most years and severe droughts (such as in the 1980s) are possible. Average precipitation for the period from 1941 to 1970 was 42.03 in. (Hunter/ESE 1990). Infrequent tropical storms are capable of delivering high winds and heavy rains to the region. The average growing season, as defined by the dates of the last and first killing frosts, is from 16 April through 2 October (Astore 1991).

The region occasionally experiences tornadoes in the spring and late fall. Hurricanes from the Atlantic Ocean usually dissipate before reaching the area, with the greatest damage caused by heavy rainfall and winds up to 40 to 50 miles per hour (Astore 1991).

3.2 HYDROGEOLOGY

3.2.1 Regional Geology

The geology of the Piedmont is mainly composed of folded metamorphic rocks with igneous intrusions of various sizes. The principal metamorphic rocks in the Piedmont Province are schist, gneiss, marble, slate, and quartzite, and the intrusive bodies are granite and diabase. Geologic structures trend from northeast to southwest. There are five isolated sedimentary basins in the province, but none impinge upon the installation (Powell and Abe 1985).

The bedrock at the installation consists of predominantly granite and granodiorite with intrusions of ferruginous quartzite. Gneiss rock formations occur along the eastern and northern sides of the installation (Commonwealth of Virginia 1993). Figure 3-2 shows the regional bedrock geology.

A layer of saprolite, a weathering product of the basement rocks, overlies the granite and metamorphic rocks throughout most of the installation. Bedrock grades upward starting with undeformed basement, then a thin zone of fractured rock, a variable layer of saprolite, and finally a thin layer of soil. The saprolite thickness and the associated bedrock surface are highly irregular; rock can be present at the ground surface or be as deep as 45 ft. Foundation borings indicate that the depth to bedrock at the installation can vary up to 30 ft within the area of a single building site (Environmental Science and Engineering, Inc. [ESE] 1982). Saprolite is generally clay-rich and of low permeability, with the clay fraction increasing upward from the parent bedrock. Saprolite developed on granite can have a higher sand fraction and somewhat higher permeability than that developed on clay-rich metamorphic rocks (e.g., slate).

Alluvial deposits of sand, silt, and clay occur in the floodplains of the principal drainages, such as the Nottoway River, Tommeheton Creek, Birch Creek, and Hurricane Branch. Where alluvium is present, it can be difficult to distinguish from saprolite.

3.2.2 Soils

The two main soil types found at Fort Pickett are the Durham-Appling-Worsham Association and the Appling-Durham-Louisburg Association. These soils typically occur in association with each other and with several other soil types. They are both members of the Red-Yellow Podzolic great soil group. These soils are formed under a deciduous, coniferous, or mixed forest

overstory, generally have a thin, gray-to-brown organic horizon (A), overlying a red, more clayey horizon (B), and red-to-brown, coarser, horizon (C) formed from the parent material. In general, the soils are either sandy silts or silty sands, varying with depth, color, and clay content (WESTON 1999a). The soils are similar in composition and all are moderately permeable (ESE 1982). Soils in the area predominantly include the Appling sandy loam with 2-7 percent slopes (Barnes 1999).

The Durham-Appling-Worsham soils are deep and well drained and develop on granite and granitic gneiss. These soils occur on broad uplands with few shallow drainages and are predominantly composed of sandy loam. Durham soil comprises approximately 65 percent of the association and occurs at moderate elevations. This soil has subsoils of brownish-yellow sandy clay loam. The Appling soil, approximately 15 percent occurrence, is composed of reddish-yellow to yellowish-red friable clay loam. The Worsham soil occurs within approximately 12 percent of the association in depressions. The remaining 8 percent is a mixture of Colfax and Cecil soils that are somewhat poorly drained (WESTON 1999a).

The Appling-Durham-Louisburg Association soils are deep and well drained, and develop on granite, granitic gneiss, and pegmatite. These soils occur on broad hills, side slopes, or along valleys. The Appling soil, found mostly on elevated areas, comprises approximately 70 percent of the association. This soil consists of reddish-yellow to yellowish-red subsoils of friable clay loam. Durham soil, approximately 12 percent occurrence, is found on lower hills. This soil has subsoils of brownish-yellow sandy clay loam. The remaining 18 percent of Louisburg soil is found on steeper slopes (WESTON 1999a).

3.2.3 Site-Specific Hydrogeology

The depth to top of bedrock is irregular being shallower on the eastern side of the site. Two downgradient wells were completed to depths of 24 and 26 ft within the overburden. Another two wells were proposed, one upgradient on the east and one on the north side of the site, but could not be completed due to auger refusal on bedrock. Auger refusal on bedrock occurred at depths ranging from 8 to 16 ft without encountering the saturated zone during several attempts to drill these wells. Well depths and groundwater gauging data are summarized in Table 3-1. The 26 October 1999 well gauging round included a water level measurement from the existing 2.5-ft diameter onsite well.

The depths to groundwater on 26 October 1999 in wells MW-1 and MW-2 were 17.35 ft and 17.74 ft, respectively (from top of casing). The existing 2.5-ft diameter water well in the former building slab had a water level of 15.35 ft below ground surface. Groundwater in these well locations was found in the overburden. The water-table elevations and groundwater gradient are shown in Figure 3-3. The groundwater gradient is 0.025 towards the northwest.

The saturated overburden aquifer at the site receives recharge from infiltration of surface precipitation. This saturated zone likely recharges the deeper bedrock aquifer, as there is likely some hydraulic connection between the overburden and bedrock. The bedrock aquifer consists of saturated fractures, or secondary porosity within the fractured granitic bedrock that has minimal primary porosity. The groundwater flow within the bedrock is strongly influenced by the orientation of the fractures; however, the average flow direction is likely towards nearby stream valleys, or a northwest direction.

3.2.4 Aquifer Characterization

Aquifer slug testing on the two monitoring wells was conducted on 28 October 1999 to calculate hydraulic conductivity (K) of the unconfined aquifer. Slug tests yielded K values on the order of 10^{-3} feet per minute in wells MW-1 and MW-2. Table 3-2 presents the calculated K values. Based on the slug tests, the average K of the unconfined aquifer was 0.001 ft/min. This average is based on the rising head test results considering that falling heads tests may be biased by effects of drainage of sand pack (Bouwer 1989).

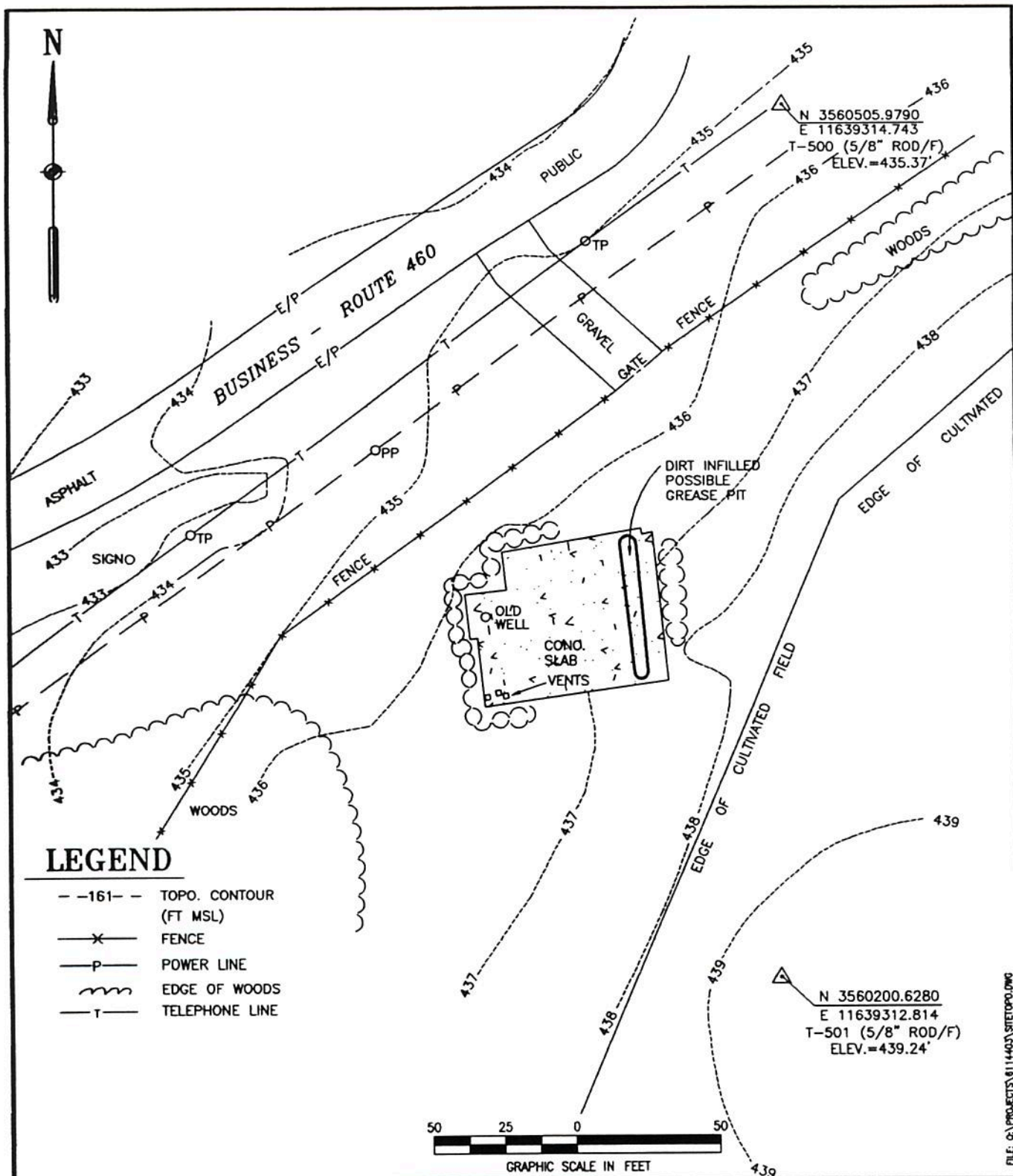
The average linear groundwater flow velocity (V) was calculated by a variation of Darcy's Law:

$$V = Ki \frac{1}{n}$$

where i is the hydraulic gradient (0.025) and n is the average fractional effective porosity (assumed 0.2). Based on these values, the groundwater flow velocity is 0.2 ft per day or approximately 70 ft/year.

3.2.5 Groundwater Use

The nearest public drinking water wells are located more than four miles from Fort Pickett. The Town of Blackstone and Fort Pickett utilize a public water system. The water source for this system is a surface water intake on the Nottoway River upstream from Fort Pickett. There are three private drinking water wells on Fort Pickett and several private wells located off the post. The Fort Pickett wells are located at Building SW101 in the southwest corner of the post several miles from PI-1, Building NW100 on the northern post boundary over 3,000 ft northeast of PI-1, and Building 4072 located near EBS-308 about 2,500 ft southeast of PI-1. This well is approximately 150 ft deep and screened within the bedrock aquifer. Another three wells located at the Virginia Polytechnic Institute (VPI) Cultural Research Center (EBS-308) are intended for research activities, but have not been used recently by VPI (WESTON 1999b). There are no private wells within 1,000 ft of PI-1, based on available information.



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SCIENCE, AND
TECHNOLOGY

PI-1
RI/FS
FT. PICKETT, VIRGINIA

SITE TOPOGRAPHY

PROJECT MGR
VAW

DESIGNED BY
SMW

DRAWN BY
JBS

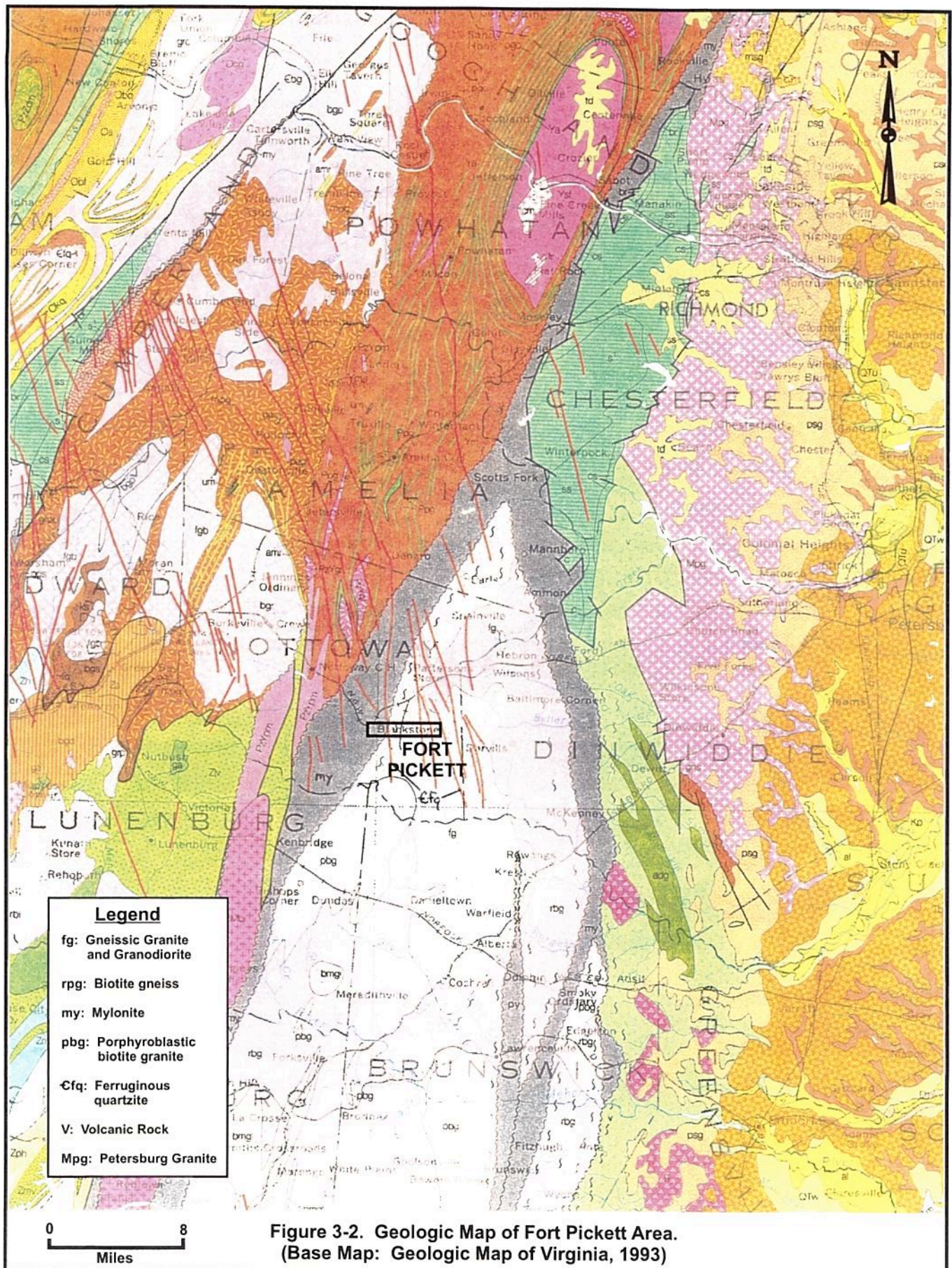
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TJP

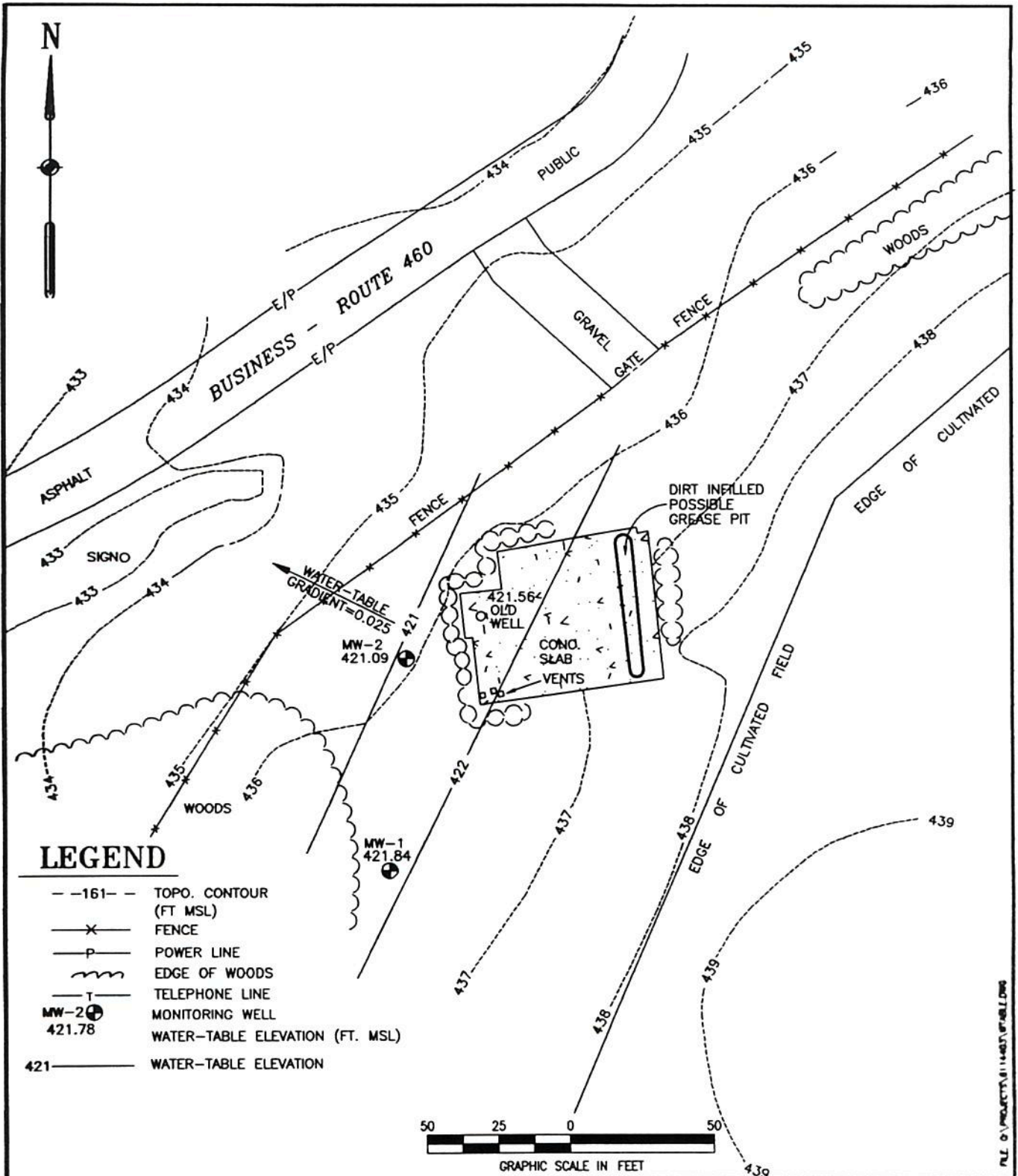
SCALE
AS SHOWN

DATE
3-9-00

PROJECT NO
61144.03

FIGURE
3-1





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TECHNOLOGY

PI-1
RI/FS
FT. PICKETT, VIRGINIA

WATER-TABLE ELEVATION
26 OCTOBER 1999

PROJECT MGR
VAW

DESIGNED BY
SMW

DRAWN BY
JBS

CHECKED BY
TJP

SCALE
AS SHOWN

DATE
3-9-00

PROJECT NO
61144.03

FIGURE
3-3

TABLE 3-1 WELL GAUGING RESULTS PI-1, FORT PICKETT, VIRGINIA

Well Number	Well Depth (ft below surface)	Casing Elevation (ft above MSL)	Gauging Date					
			10/25/1999		10/26/99		10/28/99	
			Depth to Water (ft)	Elevation	Depth to Water (ft)	Elevation	Depth to Water (ft)	Elevation
MW-1	26	439.19	17.41	421.78	17.35	421.84	17.41	421.78
MW-2	24.5	438.83	17.75	421.08	17.74	421.09	17.72	421.11
Existing Well	approx. 25	436.91			15.35	421.56		

TABLE 3-2 HYDRAULIC CONDUCTING VALUES, PI-1, FORT PICKETT

Well	Test Type	Value (ft/min)	Value (ft/day)
MW-1	Falling-head	1.76×10^{-3}	2.53
	Rising-head	1.26×10^{-3}	1.81
MW-2	Falling-head	4.05×10^{-4}	0.58
	Rising-head	1.24×10^{-3}	1.79
Average (rising head)		1.25×10^{-3}	1.8

4. NATURE AND EXTENT OF COPC

4.1 INTRODUCTION

Chapter 4 discusses the results of chemical analyses of PI-1 surface and subsurface soil, and groundwater samples. Data summary tables provided in Chapter 4 list analyte detections for each medium and undetected analytes with reporting limits greater than screening criteria. Complete data tables for each medium are included in Appendix D. An evaluation of QC sample results is discussed for each medium.

4.2 ANALYTICAL DATA QUALITY AND DATA VALIDATION

The analytical laboratory data were validated by an independent data validator, Meridian Science & Technology, Inc (MSTI). Validation procedures were in accordance with pertinent parts of the U.S. EPA Region III Modifications to the National Functional Guidelines for Inorganic Data Review (April 1993) and Organic Data Review (September 1994), along with the QA/QC requirements for the analytical methods used. Hard and electronic copies of the laboratory reports were delivered to MSTI. Upon completion of the data validation, an additional field for validation qualifier was added to the electronic database to qualify analytical results. The data summary tables in Appendix D contain validated data. Data qualified with an "R" (rejected) by the validator was excluded from use in the human health and ecological risk screenings and further evaluations. Executive summaries of the validation reports are included in Appendix E.

For cases of possible laboratory contamination, a field sample concentration had to be greater than five times the analyte value reported in the associated trip, method, field, or rinsate blank for uncommon laboratory constituents, or greater than 10 times for common laboratory constituents [acetone, methylene chloride, bis(2-ethylhexyl)phthalate, etc.] to be a valid reportable result for use in comparison to screening criteria. Concentrations of compounds in site media not meeting these criteria were considered to be artifacts of laboratory analysis and are qualified with a "B." Method blank results are included in Appendix D. QA laboratory split sample results are included in Appendix F.

Organic and inorganic compound values flagged with a "J" indicate that the reported concentration is estimated since it was detected below the RL (reporting limit). The certainty of compound identification is the same for those reported above the RL, but the actual concentration reported is estimated.

In general, the majority of analytical data were accepted after validation. The only major problem noted was low surrogate spike recovery associated with cyanide analysis of several of the soil samples. The action taken in this case was that cyanide quantitation limits for these samples were considered unusable ("R" flag). Considering that cyanide was not detected in the other 13 soil samples, as explained below, the QA laboratory split sample, and groundwater samples, it is unlikely cyanide is present at the site.

There were several minor problems with the data sets generally including issues on calibration, holding time, surrogate recovery, blank contamination (metals were detected in laboratory blanks commonly), etc. Actions ranged from the addition of qualifier flags to no action taken. The data in these cases were considered usable.

4.3 DATA EVALUATION PROCEDURES

In order to evaluate the analytical data for nature and extent of contaminants, a screening approach using established standards (Section 1.7) was used. The objective of screening the analytical results against established criteria was to evaluate whether analytes should be considered as COPC, which may warrant further evaluation through a quantitative risk assessment. Reported analytical data were reviewed according to the following criteria:

- **Surface Soil**—For ecological benchmarks: Oak Ridge National Laboratory ecological benchmarks (Efroymson et al. 1997a, 1997b) or EPA Region III BTAG ecological benchmarks (EPA 1995a) were used. Additional ecological benchmarks were obtained from and RIVM (1994, 1995, 1997). For human health risk benchmarks: RBCs Residential Use (RBC for cancer or 1/10 for non-cancer RBC) (EPA 1999a) were used. In addition, the EPA Region III (2000a) SSLs were used to evaluate potential contaminant leaching to groundwater. SSLs corresponding to a dilution-attenuation factor (DAF) of 20 were used.
- **Subsurface Soil**—RBCs Residential Use (RBC for cancer or 1/10 for non-cancer RBC) (EPA 1999a). SSLs with DAF of 20 were used.
- **Groundwater**—EPA MCLs (1996c) established under the Safe Drinking Water Act and EPA Region III, tap water RBCs (EPA 1999a). MCL and RBC values are for public drinking water supplies.

For each media, detected analyte concentrations were compared to the screening criteria. The actual criteria used are included in the media results tables. If the detected concentration exceeded a criterion or if no criteria were available, then the analyte was indicated as a COPC. Conversely, if the laboratory reporting limit exceeded the criteria, but the analyte was not detected, it was not considered as a COPC. However, these non-detected analytes were reviewed as part of nature and extent at the end of this chapter, and they were considered as uncertainties in the risk assessments. There are no regulatory or guidance criteria for TPH. TPH data, therefore, are not screened and are presented for site characterization purposes only.

Identified COPCs should be used during subsequent risk characterization and are not direct indicators of potential risks to human health and the environment. Further evaluation is required to address the potential risks. Additional data considerations for use of COPCs in the risk assessment are discussed in detail in Chapter 6. Use of background soil data (WESTON 1999a) to evaluate potential risks is included in the human health and ecological risk assessments (Chapters 6 and 7). The QA sample split results were not used to identify COPCs, but only to evaluate laboratory and sampling QA/QC.

For screening of dioxin/furan compounds, a toxicity equivalency procedure was used to screen the data. This procedure involves multiplying the dioxin/furan congener results (one-half of non-detects) by toxicity equivalency factors (TEF) (EPA 1989a) and summing these results to calculate a toxic equivalency (TEQ). This sum is compared against TEQ screening values (EPA 1989a, Ahlborg, et al. 1994). The list of TEFs is provided in Appendix D.

4.4 ANALYTICAL RESULTS

Below is a discussion of the results from samples collected from each of the matrices at PI-1. Figure 4-1 illustrates the soil and groundwater sampling locations at PI-1. In addition, each section below presents data tables listing analytes that were reported in concentrations exceeding the method detection limit or analytes for which the reporting limits were above screening criteria. Concentrations that exceeded criterion are highlighted within the tables. Results for duplicate QA/QC samples are in columns immediately following the original sample.

4.4.1 Surface Soil

A total of 9 surface soil samples, 1 surface soil duplicate, 1 USACE QA laboratory split sample, 2 equipment blanks, 2 trip blanks, and 1 field blank were collected at PI-1 on 6-7 October 1999

(Table 2-1). One of the samples, SS9-0.5 (designation for location SS-9, sample depth 0-0.5 ft), was collected in a downslope drainage swale northwest of the site, outside the former maintenance area. The remaining surface soil samples were collected from suspect locations within and around the concrete pad at PI-1. Location SB7 was within a soil filled maintenance pit that was about 6 ft deep and had a thin concrete or hard gravel base. Location SB5 was within a cylindrical hole in the concrete slab, about 8 ft deep, which may have been used for a hydraulic vehicle lift. Each of the surface soil samples was collected from the 0 to 0.5 ft below ground surface interval. Duplicate soil sample DUP was collected from SB6-0.5.

Boring SB-3 was located at an anomaly detected during the metal detector survey of the unpaved area of the site. This anomaly appeared to be approximately 3 ft long and 2-ft wide. The anomaly may be a septic tank as it is located downslope and in line with the toilet vent pipes on the slab (Figure 4-1). The associated boring SB-3 was completed adjacent to the anomaly to avoid hitting a possible hard object. The soil conditions in this boring appeared natural and not altered by the anomalous object.

The surface soil samples were analyzed for EPA TCL/TAL compounds and TPH. The surface soil samples from SB1, SB2, SB3, SB6, and SB6 duplicate were analyzed for dioxins/furans. Table 4-1 summarizes the analytical concentrations reported in the surface soil samples. Sample type "FD" refers to the duplicate sample. QA/QC sample results are shown in Table 4-2. The physical characteristics of soil samples including pH, grain size, and TOC analyses are listed in Table 4-3.

VOCs

Methylene chloride and acetone were detected in the nine samples at concentrations less than the screening criteria. These VOCs were also reported in the laboratory method blanks, field blank, or the rinsate blanks. The source of the methylene chloride is likely related to laboratory contamination and not a real presence in the surface soil. Acetone concentrations ranged from 21 to 750 µg/kg. Considering acetone was also detected in the field and rinsate blanks, its presence was likely due to sample container or laboratory contamination. However, because several samples contained acetone at levels greater than 10 times than found in the associated blanks and due to no available ecotoxicity screening value, acetone was considered a COPC.

Non-PAH SVOCs

Three SVOCs were detected at least once in five of the nine surface soil samples. Bis(2-ethylhexyl)phthalate was detected in SB6-0.5, SB7-0.5, SB8-0.5, and SB9-0.5 with concentrations ranging from 41 to 100 µg/kg. Benzyl butyl phthalate was reported in SB8-0.5 with a concentration of 120 µg/kg. The maximum reported concentrations of these phthalates were below screening criteria. Dibenzofuran was detected in SB7-0.5 and was identified as a COPC because an ecotoxicity screening value was not available.

PAHs

Seven PAHs, 2-methylnaphthalene, 1,2-benzphenanthracene, benzo[B]fluoranthene, fluoranthene, naphthalene, phenanthrene, and pyrene, were identified as COPCs for ecotoxicity in SB7-0.5. This sample was collected from fill material within the former maintenance pit. Pyrene was also an ecotoxicity COPC in SB9-0.5. 2-methylnaphthalene was identified as a COPC in SB8-0.5. There were other detections of PAHs, but the concentrations of these PAHs did not exceed RBCs or ecological benchmarks.

TPH

TPH-diesel range was detected in five samples at concentrations ranging from 35 to 160 mg/kg. No TPH-gasoline range was detected. The detected TPH may be due to residual petroleum compounds.

Pesticides and PCBs

Several pesticides were detected at trace levels below screening criteria. These pesticides included delta-HCH 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha chlordane, dieldrin, gamma-chlordane, heptachlor epoxide, endosulfan 1, endrin, and endrin ketone.

No PCBs were detected. Reporting limits for PCBs were well below the ecological benchmark of 100 µg/kg (Appendix D).

Metals and Cyanide

Twelve metals were identified as ecological COPCs in the surface soil samples. Aluminum, antimony, beryllium, chromium, copper, lead, manganese, mercury, nickel, thallium, vanadium, and zinc results exceeded ecotoxicity criteria in one or more of the nine surface soil samples. Aluminum, arsenic, and iron were identified as human health COPCs. Arsenic concentrations ranged from 0.89 to 3.2 mg/kg in the nine samples, which exceeded the RBC of 0.43 mg/kg. Aluminum concentrations ranged from 1,840 to 9,410 mg/kg. Iron concentrations ranged from 4,910 to 14,200 mg/kg.

For calcium, magnesium, potassium, and sodium, although screening criteria were not available, these metals were not considered COPCs due to their status as essential nutrients. Iron was considered an essential nutrient in the ecological assessment. This status is discussed further in Chapters 6 and 7.

No cyanide was detected; however, the detection limit was rejected in validation for five samples due to low matrix spike recoveries. Considering that cyanide was not detected in the other four samples, duplicate, and QA split sample (Appendix E), cyanide is not likely present at the site.

Dioxin/Furans

A series of dioxin/furan congeners were detected in the four tested surface soil samples, SB1, SB2, SB3, and SB6. The results for SB6-0.5 were higher in comparison to the other samples. The TEQs for the four samples exceeded the residential soil RBC of 4.3 ng/kg. Dioxin/furan is considered a human health COPC in surface soil.

QC Sample Evaluation for Surface and Subsurface Soil Samples

The QC sample results for soil samples are shown in Table 4-2. Nine metals, a dioxin congener, methylene chloride, acetone, bromodichloromethane, chloroform, and toluene were detected in certain associated method blanks (not shown), field blanks, rinse blanks, and trip blanks. Acetone, methylene chloride, and the metals were also detected in field soil samples. The low-level detections of these analytes are not considered significant in affecting the results of sample analyses. Methylene chloride and acetone are common laboratory and sampling contaminants. The VOC and metals data were not rejected for blank contamination.

The results of the laboratory split QA samples are included in Appendix F. Detected results are compared by evaluation of the relative percent difference (RPD). The RPD goal was 50 percent. This goal was exceeded for several of the analytes. Overall, there does not appear to be a laboratory bias. Differences in concentrations may be due to heterogeneity of the soil.

Slightly lower RPD values were obtained for the duplicate samples as calculated by MSTI. The validation did not require qualification of the data due to field duplicate precision bias. The validation RPD evaluation is included in Appendix E.

General Soil Characteristics

Grain size analysis, pH, TOC, and percent moisture of one surface and one subsurface soil sample from PI-1 was performed (Table 4-3). The surface soil sample for SB8-0.5 had a moisture percentage of 19.6% and a TOC of 67,300 mg/kg. The reported pH of 7.4 is typical for soil and does not indicate the presence of contamination. The relatively high TOC is likely indicative of organic matter in topsoil. The sample had a silty sand texture.

4.4.2 Subsurface Soil

The following samples were collected on 6-7 October 1999 for the subsurface soil characterization (Table 2-1): 16 subsurface soil, 1 duplicate, 1 USACE QA laboratory split sample, 2 equipment rinse blanks, 2 trip blanks, and 1 field blank. Typically, two subsurface soil samples were collected from each soil boring location; one mid-depth sample from an area of visible staining or elevated total organic vapors, and the second near the water table. The subsurface soil samples were analyzed for EPA TCL/TAL parameters and TPH. Boring SB5 was completed adjacent to the apparent former hydraulic lift structure in the concrete pad because the bottom of this structure could not be penetrated due to hard concrete. Sample SB1-26 (designation for boring SB-1, sample depth 26 ft) was analyzed for pH, TOC, and grain size. Table 4-4 summarizes the reported analytical concentrations of subsurface soil samples. Duplicate soil sample DUP2 was collected from SB1-20. QA/QC sample results are shown in Table 4-2. The physical characteristics of soil samples including pH, grain size, and TOC analyses are listed in Table 4-3.

VOCs

No VOC COPCs were identified. Acetone and methylene chloride were the only VOCs detected in the subsurface soil samples. Acetone was detected in each of the soil samples with the exception of SB6-6. Methylene chloride was reported in 14 samples with the exception of SB1-20 and SB6-6. These VOCs were also reported in the laboratory method blanks, field blank, and rinsate blanks, as explained above. The sources of these VOCs are likely related to laboratory or sample container contamination and not a real presence in site media.

Non-PAH SVOCs

There were no SVOC COPCs. SVOCs were detected in 2 of the 16 subsurface soil samples. Bis(2-ethylhexyl)phthalate was reported in SB7-6 with a concentration of 56 ug/kg. Phenol was reported in SB8-8 with a concentration of 64 ug/kg. Both of these reported concentrations did not exceed RBCs.

PAHs

No PAHs were identified as COPCs in subsurface soil. There were several reported detections of PAHs, especially in SB7-6, but concentrations did not exceed RBC screening criteria.

TPH

TPH-diesel range was detected in two samples, SB5-4 and SB7-6, at concentrations ranging from 94 to 390 mg/kg. No TPH-gasoline range was detected. The detected TPH may be due to residual petroleum compounds within the fill material at these locations.

Pesticides and PCBs

No pesticides or PCBs were identified as COPCs in the subsurface soil. Pesticides were detected in 2 of the 16 samples. 4,4'-DDD and endosulfan II were reported in SB5-4 with concentrations of 0.87 µg/kg and 1.4 µg/kg, respectively. 4,4'-DDD and 4,4'-DDE were reported in SB7-6 with concentrations of 8.1 µg/kg and 12.0 µg/kg, respectively.

Metals and Cyanide

Seven TAL metals were identified as COPCs in the subsurface soil samples. Arsenic, aluminum, chromium, iron, manganese, thallium, and vanadium were detected in the subsurface soil samples with concentrations exceeding RBC screening criteria. Arsenic concentrations ranged from 0.61 to 2.2 mg/kg. The residential soil RBC for arsenic is 0.43 mg/kg. Chromium was detected at 33.1 mg/kg in SB4-10, exceeding the Cr^{+6} RBC screening value of 23 mg/kg. Iron was reported in the 16 subsurface soil samples with concentrations ranging from 1,990 mg/kg to 54,900 mg/kg. During subsurface soil sampling, iron oxides were observed at depths ranging from 4 to 20 ft below ground surface.

No cyanide was detected; however, the quantitation limit was rejected in validation for six samples due to low matrix spike recoveries. Considering that cyanide was not detected in the other four samples, duplicate, and QA split sample SB6-15.5 (Appendix F), cyanide is not likely present at the site.

General Soil Characteristics

Grain size analysis, pH, TOC, and percent moisture of one surface and one subsurface soil sample from PI-1 was performed (Table 4-3). Subsurface soil sample SB1-26 was collected within the unconfined aquifer. Total organic carbon was not detected. The sample had a silty sand texture.

4.4.3 Groundwater Sampling

A groundwater sample was collected from each of the two installed monitoring wells at PI-1 (Figure 4-1). Groundwater sampling occurred on 26 October 1999 using the low-flow protocol. Groundwater samples were analyzed for EPA TCL/TAL parameters and TPH. In addition to the two groundwater samples, one duplicate sample, one USACE QA laboratory sample, one equipment blank, one field blank, and one trip blank were submitted for analysis. Duplicate MW-DUP was collected from MW-1. Table 4-5 summarizes the analytical results for the groundwater samples collected from PI-1. QA/QC sample results are shown in Table 4-6. Groundwater quality parameters measured during purging are summarized in Table 4-7.

VOCs

No VOC COPCs were identified. No VOCs were detected in the two groundwater samples and duplicate.

Non-PAH SVOCs and PAHs

Bis(2-ethylhexyl)phthalate was detected in MW-2 at 1.6 ug/L, which did not exceed screening criteria. For several SVOCs and PAHs, RBCs were below detection limits but the compounds were not detected. These compounds are reviewed at the end of this chapter.

Pesticides and PCBs

Four pesticides were identified as COPCs in groundwater. Aldrin, alpha-HCH, heptachlor, and heptachlor epoxide were detected with concentrations that exceeded RBC screening criteria. Aldrin was reported in MW-1 and MW-2 with concentrations of 0.0073 and 0.07 µg/L, respectively. The RBC for aldrin is 0.0039 µg/L. Alpha-HCH was reported in MW-2 with a concentration of 0.13 ug/L, above the RBC of 0.011 ug/L. The MW-1 normal sample and its corresponding duplicate contained traces of heptachlor epoxide (0.01 ug/L) and heptachlor was detected in MW-2 at 0.08 ug/L. 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endosulfan sulfate, gamma-HCH, and endrin ketone were also detected, however, concentrations did not exceed screening criteria.

For PCBs, the RBCs were below detection limits, but non-detected PCBs were not considered COPCs. PCBs were not detected in the soil at the site, and therefore, are not likely present in groundwater.

Metals and Cyanide

Total and dissolved TAL metals were analyzed for MW-2 due to elevated turbidity readings during purging. The dissolved results were used to evaluate COPC considering the significantly higher concentrations observed in the non-filtered sample. Based on EPA (1992a) guidance concerning use of filtered data, when there is a notable difference between filtered and non-filtered data, then filtered results should be used. Only manganese was identified as a COPC for groundwater sample MW-2, where both total and dissolved concentrations exceeded the RBC. Total aluminum, chromium, and iron concentrations in MW-2 exceeded RBCs; however, the

dissolved concentrations did not exceed RBCs and, therefore, these metals are not COPCs. No cyanide was detected.

QC Sample Evaluation for Groundwater Samples

The QC sample results for groundwater samples are shown in Table 4-6. Nine metals, methylene chloride, bromodichloromethane, and chloroform were detected in certain associated method blanks (not shown), field blanks, rinse blanks, and trip blanks. These VOCs were not detected in the well samples. The low-level detection of the metals in samples was not considered significant in affecting the results of sample analyses. The metals data were not rejected for blank contamination.

The results of the laboratory split QA samples are included in Appendix F. Detected results are compared by evaluation of the RPD. The RPD goal was 30 percent. This goal was exceeded for potassium. Several pesticides were detected in the field sample at trace levels, but not in the split sample even though detection limits were similar. Overall, there does not appear to be a laboratory bias.

Slightly lower RPD values were obtained for the duplicate groundwater samples as calculated by MSTI. The validation did require qualification of pesticide analytes that indicated a field duplicate precision bias. In addition, for some metals, dissolved concentrations in MW-2 were higher than total concentrations. Affected analytes were qualified as estimated. The validation RPD evaluation is included in Appendix E.

Groundwater Quality Parameters

The dissolved oxygen and pH of the well samples MW-1 and MW-2 were 3.64 and 4.87 mg/L, and 4.36 and 4.79, respectively (Table 4-7). These values are not indicative of contaminated groundwater.

4.5 SUMMARY OF THE NATURE AND EXTENT OF CONTAMINATION

4.5.1 Identified COPCs

As discussed in Section 4.3, certain pesticides and PAHs, acetone, dibenzofuran, dioxins, and several of the TAL metals were considered to be COPCs for certain sampled media. Acetone

and dibenzofuran were identified as COPCs in surface soil due to the lack of ecological screening criteria. These COPCs, although not indicative of a potential risk, would conservatively be included in an ecological risk characterization.

Other analytes were identified as COPCs in media due to exceedance of risk-based criteria. The identified COPCs, their concentration range, and the number of sample locations where the concentration exceeded the screening criteria for each medium sampled are listed in Table 4-8. The media are listed in order from potential source areas onsite, such as surface soil locations, along potential pathways of contaminant transport, such as subsurface soil and groundwater, and to downgradient surface soil. A potential contaminant pathway link and the extent of contaminants can be visualized by following a certain contaminant from a potential source area to downgradient areas. For example, potential pathways could include from onsite surface soil to offsite surface soil, or onsite subsurface soil to groundwater.

Within surface soil, PAH COPCs predominantly were located at sample SB-7-0.5, which was within the former maintenance pit. This soil also contained dibenzofuran as a COPC. These elevated hydrocarbons are likely due to the nature of the fill material placed in the pit. The elevated diesel range-TPH in several samples reflects the presence of heavy hydrocarbons. Metal concentrations, however, were comparable to other surface and subsurface soil at the site. One PAH, 2-methylnaphthalene was a COPC in SB-8-0.5 located next to the concrete pad. Dioxin was a COPC in SB6-0.5 also located adjacent to the pad. These COPCs and the TPH may reflect residual contamination from former vehicle repair operations at the site. The presence of acetone in the surface and subsurface soil samples is likely due to laboratory or sampling effect, but was considered as an ecological COPC.

Considering the 14 COPC metals, in general metal concentrations were at similar levels in surface soil across the site and may reflect background concentrations (Table 1-2). However, the lead level of up to 118 mg/kg and zinc up to 181 mg/kg may be due to site conditions. The seven metal COPCs in subsurface soil, likewise, may reflect background conditions and not due to site impacts. Background soil quality is discussed further in Section 6.7.

In groundwater samples, the four pesticides COPCs may reflect a ubiquitous presence in the general area. It is known that pesticides were routinely sprayed in the area by former Fort Pickett training operations (Section 1.5). Pesticide levels in shallow soil at the site were negligible, which indicated that there was not a significant pesticide source onsite. The elevated

manganese level in groundwater may be due to the relatively acidic groundwater (pH 4-5) mobilizing this metal from the soil.

The downslope surface soil sample collected from the swale did not appear to be significantly impacted by the site. The COPCs pyrene, iron, lead, and zinc in this sample may, however, be associated with the site conditions.

4.5.2 Analyte Reporting Limits Above Screening Criteria

Within soil and groundwater, the analytes that were not detected, but had detection limits above the screening criteria are listed in Table 4-9. In surface and subsurface soil, n-nitrosodi-n-propylamine had a reporting limit greater than the RBC. This compound is reported to be a contaminant in the pesticide s-ethyl dipropylthiocarbamate (EPTC) (Briggs 1992) and, therefore, there is a possibility that it may be present at the site due to past, general pesticide use. Considering that the detected pesticide concentrations were one to two orders of magnitude less than the detection limit of this SVOC, EPTC was not reported to be used at Fort Pickett, and this SVOC was not detected in the nine surface and 16 subsurface soil samples and duplicates, this possibility of presence is considered negligible.

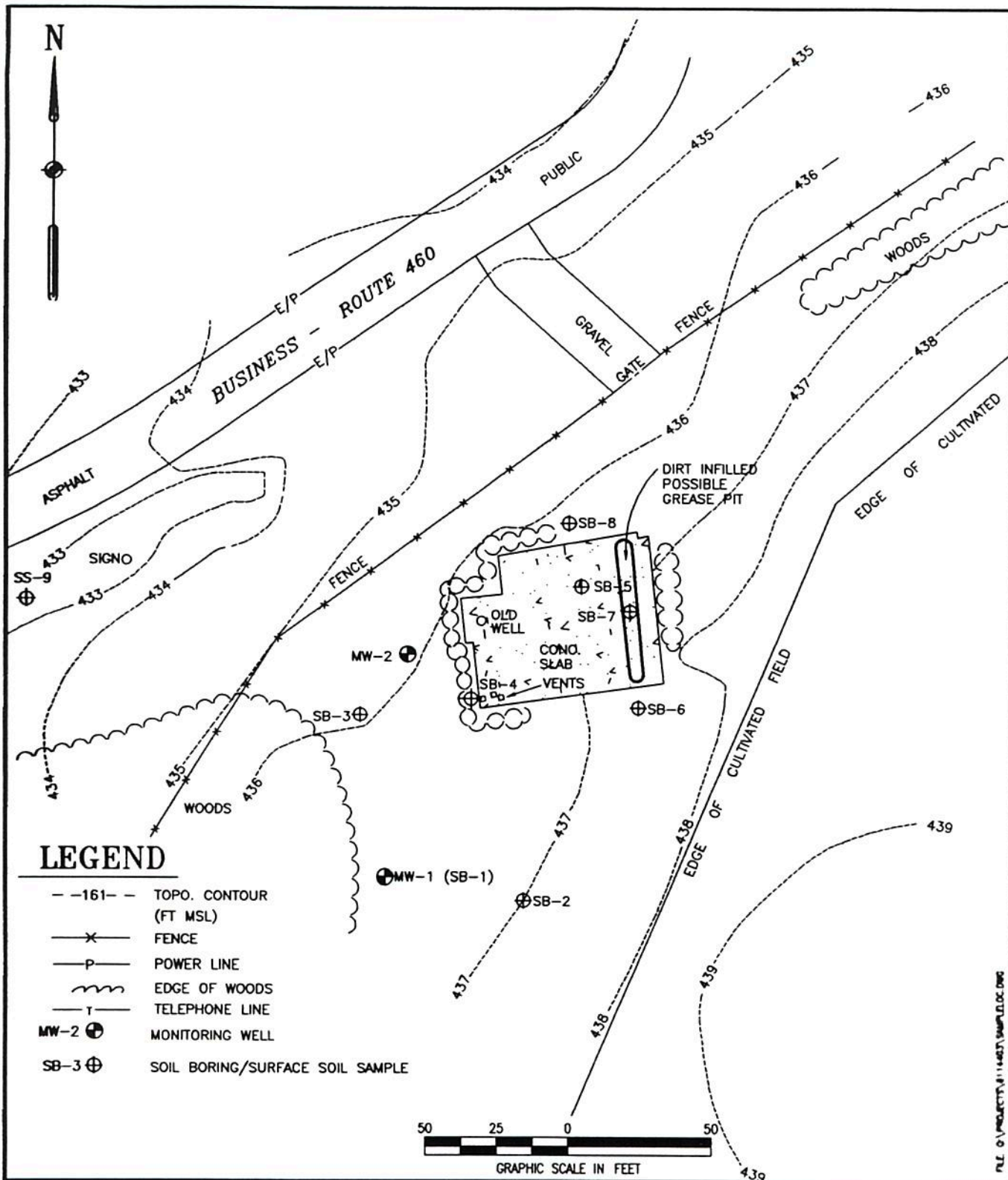
Within groundwater, the detection limits for 10 VOCs and 12 SVOCs were above the RBC screening criteria as shown in Table 4-5. The VOCs included a series of chlorinated hydrocarbons, which are known solvent compounds or can be associated with solvents in manufacturing or degradation products (Montgomery and Welkom 1990). Solvents may have been used at the site. Considering that these chlorinated compounds were not detected in soil or groundwater samples at the site, except for TCE detected below the RBC in three shallow soil samples collected by WESTON (1998), the possibility of the presence of these VOCs is considered negligible. The listed SVOCs can be associated with insecticides, hydraulic fluid, or manufacturing processes (Montgomery and Welkom 1990), which may have been used at the site.

Similarly for benzo[a]pyrene and dibenz[a,h]anthracene in groundwater, the RBCs were below detection limits. These compounds can be associated with combustion of fuels. The RBC for the pesticide toxaphene, 0.061 µg/L, was below the detection limit of 0.09 µg/L. Also, the RBCs were below detection limits for PCBs, arsenic, and thallium. It is possible that arsenic and thallium may be present in groundwater considering their detection in soil at the site. For the PCBs and toxaphene, the compounds were not detected in the collected surface and subsurface

soil samples and, therefore, they are not likely present in groundwater. The detected concentrations of the two PAHs in soil were one to three orders of magnitude less than the SSLs. Therefore, the presence of these PAHs in groundwater is considered negligible.

4.5.3 Analyte Concentrations Exceeding SSLs

The SSLs are listed with analyte results in Tables 4-1 and 4-4. Within surface and subsurface soil, the concentrations of naphthalene and arsenic were above the SSLs for migration to groundwater. The higher naphthalene concentrations were detected in soil collected from the filled maintenance pit. The bottom of this pit has a hard concrete or compacted layer, which tends to trap percolated water. Thus, the potential migration of naphthalene into groundwater below this pit may be reduced. Arsenic was present in soil across the site and could potentially leach to groundwater. The thallium concentration was above the SSL in one subsurface sample. Arsenic and thallium soil concentrations may reflect background conditions. Dioxin concentrations (TEQ) in surface soil were above SSLs and, therefore, could potentially leach to groundwater. Lastly, the detection limit of n-nitrosodi-n-propylamine (not detected) was above SSLs for surface and subsurface soil samples. None of these compounds were detected in groundwater. Dioxins were not tested in groundwater.



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PI-1
RI/FS
FT. PICKETT, VIRGINIA

EA SAMPLE LOCATIONS

PROJECT MGR VAW	DESIGNED BY SMW	DRAWN BY JBS	CHECKED BY TJP	SCALE AS SHOWN	DATE 3-9-00	PROJECT NO 61144.03	FIGURE 4-1
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TABLE 4-1 SURFACE SOIL ANALYTE DETECTIONS AND COPCS, PI-1, FORT PICKETT

Sample Location: Depth(ft): Sample Date:															
C/N	Analyses	SSL (soil to gw)	Resid. Soil RBC	Surface Soil Ecotox	Reporting Limit	SB1 10/7/1999	SB2 10/7/1999	S B3 10/7/1999	SB4 10/7/1999	SB5 10/6/1999	SB6 10/7/1999	SB6-DUP 10/7/1999	SB7 10/6/1999	SB8 10/6/1999	SB9 10/6/1999
VOCs (ug/kg)															
C	METHYLENE CHLORIDE	19	85,000	300	2	2 B	4 B	4 B	4 B	5 B	3 B	3 B	4 B	3 B	4 B
N	ACETONE	2,500	7,800,000		2	21 B	120 J	100 J	150 J	750 J	42 B	29 B	130	190 L	110 B
N	M & P XYLENES	250,000	160,000,000	100	1									5 L	
N	O-XYLENE	230,000	160,000,000	100	1									4 L	
TPH (ug/kg)															
TPH, DIESEL-RANGE									69,000	160,000	35,000	38,000	45,000	62,000	
SVOCs (ug/kg)															
C	BIS(2-ETHYLHEXYL) PHTHALATE	2,900,000	46,000	30,000	56	99 U	99 U	95 U	110 U	100 J	74 B	41 B		53 J	56 J
C	N-NITROSODI-N-PROPYLAMINE	0.047	91		100	360 U	360 U	340 U	380 U	420 UJ	330 U	330 U	340 UJ	390 U	370 U
C	PENTACHLOROPHENOL		5,300	3,000	330	150 U	150 U	140 U	160 U	180 U	140 U	140 U	160 U	160 U	150 U
N	2,4-DIMETHYLPHENOL	6,700	1,600,000	30,000	140	720 U	720 U	700 U	780 U	860 UJ	670 U	670 U	690 U	790 U	740 U
N	2,4-DINITROPHENOL		160,000	20,000	670	180 U	180 U	180 U	200 U	220 U	170 U	170 U	180 U	200 U	190 U
N	4-METHYLPHENOL		390,000	30,000	170									120	
N	BENZYL BUTYL PHTHALATE	17,000,000	16,000,000	30,000	70								180		
N	DIBENZOFURAN	7,700	310,000		80										
PAHs (ug/kg)															
C	1,2-BENZOPHENANTHRACENE		88,000	100	1.8	5.3	4.7 J	11	31 J	30 J	25 J	21	130 J	17 J	76 J
C	BENZ[A]ANTHRACENE	1,500	880	100	1.8	2	2.1 J	4.7	18 J	15 J	15 J	20	87 J	6.2 J	44 J
C	BENZO[A]PYRENE	370	88	100	1.8	2	1.9	4.1	12 J	17 J	10 J	18	85 J	8.2 J	43 J
C	BENZO[B]FLUORANTHENE	4,500	880	100	1.8	5.8	4.4	8.7	32 J	40 J	26 J	52	150 J	24 J	95 J
C	BENZO[K]FLUORANTHENE	45,000	8,800	100	1.8	2.5			5.1 J	10 J	14 J	12	41 J	5.2 J	31 J
C	DIBENZ[A,H]ANTHRACENE	1,400	88	100	1.8				6 J			3.3	20 J		13 J
C	INDENO[1,2,3-CD]PYRENE	13,000	880	100	1.8		2.3	3.5	9.1 J	17 J	9.1 J	18	62 J	11 J	40 J
N	2-METHYLNAPHTHALENE	22,000	1,600,000	100	1.8	5.6	6.3	33	73 J	49 J	50	56	660	160	74 J
N	ACENAPHTHENE	100,000	4,700,000	100	1.8			1.9	6.4 J	4.7 J	2.4	5.8	12 J		3.6 J
N	ACENAPHTHYLENE		2,300,000	100	1.8				2.9 J	2.1 J	2.5	3.6	14 J		9.9 J
N	ANTHRACENE	470,000	23,000,000	100	1.8				3.1 J	2.9 J	2.7	5.5	19 J	2.4 J	8.5 J
N	BENZO[G,H,I]PERYLENE		2,300,000	100	1.8		2.4	5.5	13 J	19 J	11 J	14	77 J	15 J	50 J
N	FLUORANTHENE	6,300,000	3,100,000	100	1.8	3.3	2.8	6.2	21 J	34 J	9.5	15	110 J	10 J	62 J
N	FLUORENE	140,000	3,100,000	100	1.8				5.4 J	9.1 J	3.4	5.1	26 J	2.9 J	6.7 J
N	NAPHTHALENE	150	1,600,000	100	1.8	4.5	6	22	46 J	18 J	42	51	270 J	49 J	46 J
N	PHENANTHRENE		2,300,000	100	1.8	9	8.8	28	79 J	43 J	34	33	360 J	23 J	96 J
N	PYRENE	680,000	2,300,000	100	1.8	4.5 J	4.1 J	8	38 J	74 J	14 J	12	270 J	23 J	120 J
Pesticides/PCBs (ug/kg)															
C	4,4'-DDD	11,000	2,700	100	0.45				3.6 J		0.6 J	1.4 J	8.7 J		0.79 J
C	4,4'-DDE	35,000	1,900	100	0.42				2.9 J		1.6 J	1.8 J	31 J	1.7 J	1.4 J
C	4,4'-DDT	1,200	1,900	100	0.81				8.5 J				40 J	1.5 J	
C	ALPHA-CHLORDANE	920	1,800	100	0.77								3.4		
C	DELTA-HCH		350	1000	0.61									0.89 J	
C	DIELDRIN	2.2	40	100	0.46				1.1 J		0.84	0.74	1.3 J		

TABLE 4-1 (Continued)

C/N Analytes	SSL (soil to giv)	Resid. Soil RBC	Surface Soil Ecotox	Reporting Limit	Sample Location: Depth(ft): Sample Date:									
					SB1 10/7/1999	SB2 10/7/1999	SB3 10/7/1999	SB4 10/7/1999	SB5 10/6/1999	SB6 10/7/1999	SB6-DUP 10/7/1999	SB7 10/6/1999	SB8 10/6/1999	SB9 10/6/1999
C GAMMA-CHLORDANE	920	1,800	100	0.38	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
C HEPTACHLOR EPOXIDE	25	70	100	0.89				1.1 J				2.7 J		
N ENDOSULFAN I	20,000	470,000	25,000	0.89				1.8					2.7 J	
N ENDRIN	5,400	24,000	100	1.9										1.3
N ENDRIN KETONE		23,000	30	0.82										
Metals (mg/kg)														
CALCIUM				6.1	684	79,100*	49,900*	4,220*	5,790	162,000	189,000*	1,810	127,000	8,250
MAGNESIUM				10.3	253 J	14,000 J	15,700 J	690 J	993	455,000 J	50,200 J	946	51,200	2,880
POTASSIUM				23.2	232	751 J	709 J	464 J	1,450	786 J	515 J	392	576	434
SODIUM				10.5	85.5 B	129 B	127 B	121 B	133 B	181 B	178 B	102 B	216 B	115 B
C ARSENIC	0.026	0.43	10	0.09	1.1 J	1.6 U	1.6 U	3.1 U	4.290*	2.9 L	3.2 L	0.89 J	2.5 J	2.1 U
N ALUMINUM		78,000	1	1.9	4,190	5,240 J	4,540 J	9,410 J	4,290*	4,080 J	4,350 J	2,130*	1,840*	4,850*
N ANTIMONY	13	31	0.48	0.22	0.27 L	35 J	58.8 J	40.4 J	30.2	77 J	76.8 J	20.2	47.7	48.3
N BARIUM	2,100	5,500	440	0.91	34.3	0.24 J	0.3 J	0.29 J	0.17	0.27 J	0.21 J	0.22	0.2	0.3
N BERYLLIUM	1,200	160	0.02	0.01	0.23 J	0.06 B	0.17 B	0.42	0.15 B	0.7	0.67	0.25	1	1
N CADMIUM	27	39	2.5	0.02	4.6	5.2 J	9.6 J	9.2 J	9	7.8 J	5.9 J	5.7	7.1	10
N CHROMIUM (as Cr+6)	42	230	0.0075	0.09	0.85 L					2.1 L	1.9 L		1.2 L	0.93 L
N COBALT		4,700	20	0.27	1.4 J	3.9 J	3.8 J	5.9 J	7.8	16 J	21.1 J	4.2	33.1	11.2
N COPPER	11,000	3,100	15	0.08	1.4 J	6,750 J	6,480 J	14,200 J	7,770	10,100 J	18,900 J	5,800	11,000	11,500
N IRON		23,000		0.86	11.5	13.5 J	22.4 J	118 J	51.2	75.2 J	92.3 J	84.1	114	118
N LEAD (EPA action level)	400		0.01	0.09	42.3	64 J	131 J	95.3 J	86.7 L	87 J	73.8 J	89.1 L	117 L	105 L
N MANGANESE	950	1,600	100	0.28		0.06 U		0.06	0.07 U				0.1	0.06
N MERCURY (as HgCl2)		24	0.058	0.04	1.1	2.7	2.8	3.6		7.8	8.7		9.1	2.2 L
N NICKEL		1,600	2	0.26		0.37 B		0.32 B	0.61 B	0.19 B		0.14 B	0.35 B	0.35 B
N SELENIUM	19	390	1	0.14	0.18 U	0.18 U	0.18 U	0.2 U	0.22 U	0.17 U	0.17 U	0.17 U	0.19 U	0.18 U
N SILVER	31	390	2	0.17	0.17	0.15	0.14	0.19	0.14 U	0.11 U	0.11 U	0.11 U	0.12 U	0.23
N THALLIUM	3.6	5.5	0.001	0.11	8.4	9.3*	14.8*	18.2*	11*	9.5*	7.1*	5.4*	7.6*	13.2*
N VANADIUM	5,100	550	0.5	0.74	4.1 J	9.3 J	24.7 J	98.1 J	55.9	80.5 J	56.4 J	37.3	181	124
N ZINC	14,000	23,000	10	0.26	0.24 UR				0.29 UR			0.23 UR	0.26 UR	0.25 UR
Inorganics (mg/kg)														
N CYANIDE	150	1,600		0.23										

BLANK CELL INDICATES ANALYTE NOT DETECTED.

ECOLOGICAL COPC: Light Shade and Bold Type: Analyte was detected above ecological benchmark, or detected and benchmark was not available.
HUMAN HEALTH COPC: Dark Shade and Bold Type: Analyte was detected above screening benchmark.
C/N: Carcinogenic or noncarcinogenic analyte. Full value RBC shown for all analytes. "N" analyte compared to 1/10 of RBC. Selected metal species for RBC indicated.

*: Duplicate analysis is not within control limits.

R: Unusable result.

B: Not detected substantially above level in laboratory or field blank.

J: Estimated value or quantitation limit.

U: Not detected.

TABLE 4-1 (Continued)

C/N	Analytes	SSL (soil to gw)	Resid. Soil RBC	Surface Soil Ecotox	Sample Location: Depth(ft): Sample Date:	Reporting Limit	SB1	SB2	SB3	SB4	SB5	SB6	SB6-DUP	SB7	SB8	SB9
							0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
	Dioxins (ng/kg)						4,160 J	1,970	1,540			1,450	1,450			
	1,2,3,4,6,7,8,9-OCTACHLORODIBENZODIOXIN					0.14	61.8	44.4	28.6			47.2	45.8			
	1,2,3,4,6,7,8-HEPTACHLORODIBENZO-P-DIOXIN					0.08	3.93 J	2.49 J	4.29 J			27.2 J	28.6 J			
	1,2,3,4,6,7,8-HEPTACHLORODIBENZOFURAN					0.1	0.34	0.22	0.33 J			2.7	2.32			
	1,2,3,4,7,8,9-HPCDF					0.1	0.82	0.79	0.47			1.33	1.26			
	1,2,3,4,7,8-HEXACHLORODIBENZO-P-DIOXIN					0.2	1.14	0.55	0.65			6.2	8.1			
	1,2,3,4,7,8-HEXACHLORODIBENZO-P-DIOXIN					0.09	2.42	2.43	1.38			6.87	6.35			
	1,2,3,6,7,8-HEXACHLORODIBENZO-P-DIOXIN					0.18	2.67	1.4	2.48			24.9	28.4			
	1,2,3,6,7,8-HXCDF					0.23	0.8	0.39 J								
	1,2,3,7,8,9-HXCDF					0.04	0.61	0.64	0.37			1.89	1.93			
	1,2,3,7,8-PENTACHLORODIBENZO-P-DIOXIN					0.08	0.47	0.34	0.4			3.55	3.64			
	1,2,3,7,8-PENTACHLORODIBENZOFURAN					0.21	6.37	3.24	5.68			62.3	66.7			
	2,3,4,6,7,8-HXCDF					0.07	16.1	8.37	12.6			156	164			
	2,3,4,7,8-PCDF					0.05	0.92	0.75	0.8			9.02	10.3			
	2,3,7,8-TETRACHLORODIBENZOFURAN					0.08	4.26	1.67	3.67			9.87	10.3			
	OCDF						4,710	2,320	1,990			5,670	6,340			
	PCDD/FS						14.9	8.27	9.62			93.1	98.3			
	TEQ (ND=0)						15	8.31	9.62			93.1	98.3			
	TEQ EMPC (ND=0)						15	8.31	9.69			93.4	98.5			
	TEQ EMPC (ND=1/2)						15	8.31	9.69			100 J	98.5 J			
	TOTAL HPCDD					0.08	142 J	99.9 J	68.5 J			66.7 J	67.2 J			
	TOTAL HPCDFS					0.1	10.7 J	5.29 J	9.24 J			62.5 J	60.1 J			
	TOTAL HXCDD					0.1	12.8 J	20.4 J	12.7 J			754 J	805 J			
	TOTAL HXCDFS					0.23	77.1 J	37.1 J	66.1 J			25.2 J	26.4 J			
	TOTAL HXCD					0.04	3.54 J	4.86 J	2.52 J			2210 J	2650 J			
	TOTAL PCDD					0.08	206 J	118 J	191 J			2.63 J	4.39 J			
	TOTAL PCDFS					0.06	0.43 J	0.62 J	0.17 J			991 J	1,170 J			
	TOTAL TCDD					0.05	92.6 J	65.8 J	98.2 J			3.48	3.28			
	TOTAL TCDFS					0.09	1.54	1.62	0.87			0.37	0.44			
	1,2,3,7,8,9-HEXACHLORODIBENZO-P-DIOXIN					0.06	0.17 J	0.22	0.1 U			93.4	98.5			
	2,3,7,8-TCDD						14.9	8.28	9.69			93.4	98.5			
	TEQ (ND=1/2)		8.6	4.3	10,000		14.9	8.28	9.69			93.4	98.5			

HUMAN HEALTH COPC: Shaded and bold type: Analyte was detected above screening benchmark, or detected and benchmark not available.
J: Estimated value or quantitation limit.
C/N: Carcinogenic or noncarcinogenic analyte. Full value RBC shown for all analytes.
U: Not detected.

TABLE 4-2 SOIL SAMPLING FIELD QA/QC BLANK RESULTS, PI-1

Field Sample ID:	PI1-FIELD-1	EBS79-PI1-RB-DP3	PI1-RB-DP4	EBS79-PI-RB-HSA1	TRIP BLANK4	TRIP BLANK 6
Sample Type:	FB	RB	RB	RB	TB	TB
Associated Matrix:	SOIL	SOIL	SOIL	SOIL	SOIL	SOIL
Sample Date:	10/7/1999	10/6/1999	10/7/1999	10/7/1999	10/6/1999	10/7/1999
Inorganic (mg/L)						
CYANIDE	0.01 UL	0.01 U	0.01 UL	0.01 UL		
Metals (ug/L)						
ALUMINUM	37.9 U	37.9 U	37.9 U	37.9 U		
ANTIMONY	1 U	1 U	1 U	1 U		
ARSENIC	1.7 U	1.7 U	1.7 U	1.7 U		
BARIUM	12.3 U	12.3 U	12.3 U	12.3 U		
BERYLLIUM	0.4 UL	0.4 UL	0.4 UL	0.4 UL		
CADMIUM	0.23 B	0.2 U	0.2 U	0.2 U		
CALCIUM	67.5 B	55.6 U	186 B	106 B		
CHROMIUM	0.7 UL	0.7 UL	0.7 UL	0.7 UL		
COBALT	7.5 UL	7.5 UL	7.5 UL	7.5 UL		
COPPER	2.1 UL	2.1 UL	2.1 UL	2.1 UL		
IRON	5.5 UL	5.5 UL	5.5 UL	16.3 B		
LEAD	1.3 B	1.7 B	1.4 B	1.4 B		
MAGNESIUM	95.6 U	95.6 U	95.6 U	95.6 U		
MANGANESE	2.4 UL	2.4 UL	2.4 UL	2.4 UL		
MERCURY	0.1 U	0.1 U	0.1 U	0.1 U		
NICKEL	13.1 U	13.1 U	13.1 U	13.1 U		
POTASSIUM	174 U	174 U	174 U	174 U		
SELENIUM	1.8 U	1.8 U	1.8 U	1.8 U		
SILVER	2.2 U	2.2 U	2.2 U	2.2 U		
SODIUM	700 B	636 B	694 B	726 B		
THALLIUM	1.7 U	1.7 U	1.7 U	1.7 U		
VANADIUM	5.6 U	5.6 U	5.6 U	5.6 U		
ZINC	1.7 UL	1.7 UL	1.7 UL	1.7 UL		
Dioxin/Furans (ng/L)						
1,2,3,4,6,7,8,9-OCTACHLORODIBENZODIOXIN	0.00672 B		0.0097 B	0.0064 B		
TEQ (ND=1/2)	0.00109		0.00189	0.001		
TEQ EMPC (ND=0)	0		0	0		
TEQ EMPC (ND=1/2)	0.0011		0.0019	0.001		
SVOC (ug/L)						
BIS(2-ETHYLHEXYL) PHTHALATE	1.2 J	2 U	2 U	2 U		
VOC (ug/L)						
ACETONE	6	11 J	6	6	3 U	3 U
BROMODICHLOROMETHANE	2	1	1	1	0.4 U	0.4 U
CHLOROFORM	31	23	25	26	0.2 U	12
METHYLENE CHLORIDE	0.5 U	0.5 UJ	0.5 U	0.5 U	0.5 UJ	0.5 U
TOLUENE	0.5 U	0.5 U	0.5 U	0.3 J	0.5 U	0.5 U

B: Not detected substantially above level in laboratory or field blank.
J: Estimated value or quantitation limit.
L: Value or quantitation limit may be higher.
U: Not detected.

FB: Field blank
RB: Rinse blank
TB: Trip blank

TABLE 4-3 PHYSICAL CHARACTERISTICS OF SOIL, PI-1, FORT PICKETT

Parameter	Sample Designation	
	PI-1-SB8-0.5	PI-1-SB1-26
% Moisture	19.6	N/A
pH	7.4	N/A
Total Organic Carbon (TOC) (mg/kg)	67300	3760 U
Particle Size		
% Gravel	11.9	1.1
% Sand	56.4	75.2
% Silt	20.2	17.1
% Clay	11.5	6.6
Material Description	Silty SAND	Silty SAND
USCS Classification	SM	SM

N/A: Not analyzed

U: Not detected.

TABLE 4-4 SUBSURFACE SOIL ANALYTE DETECTIONS, PI-1, FORT PICKETT

C/N	Analytes	SSL (cell to g ^m)	Reid. Soil RBC	Sample Location		SB1 10/7/1999	SB1-DUP 10/7/1999	SB2 10/7/1999	SB2 10/7/1999	SB3 10/7/1999	SB3 10/7/1999	SB4 10/7/1999	SB4 10/7/1999	SB5 10/6/1999	SB5 10/6/1999	SB6 10/7/1999	SB7 10/6/1999	SB7 10/6/1999	SB8 10/6/1999	SB8 10/6/1999
				Depth (ft)	Reporting Limit															
C	Volatiles (g/g)																			
C	METHYLENE CHLORIDE	19	85,000	2																
N	ACETONE	2,500	7,800,000	2																
TPH (g/g)																				
TPH DIESEL RANGE																				
Semi-Volatiles (g/g)																				
C	BIS(2-ETHYLHEXYL) PHTHALATE	2,900,000	46,000	63																
N	DIBENZOFURAN	7,700	310,000	86																
N	N-NITROSO-DI-N-PROPYLAMINE	0.047	91	100																
N	PHENOL	130,000	47,000,000	81																
PAHs (g/g)																				
N	2-METHYLNAPHTHALENE	22,000	1,600,000	19																
C	1,2-BENZOPHENANTHRACENE	88,000	88,000	19																
C	BENZO[ANTHRACENE	1,500	880	22																
C	BENZO[APYRENE	370	88	22																
C	BENZO[FLUORANTHENE	4,500	880	19																
C	BENZO[K]FLUORANTHENE	45,000	8,800	22																
C	DIBENZO[A,H]ANTHRACENE	1,400	88	22																
C	INDENOL[2,3-CD]PYRENE	13,000	880	22																
N	ACENAPHTHYLENE	100,000	4,700,000	21																
N	ACENAPHTHENE	470,000	23,000,000	22																
N	ANTHRACENE	23,000,000	23,000,000	22																
N	BENZO[G,H]PERYLENE	63,000,000	3,100,000	19																
N	FLUORENE	140,000	3,100,000	21																
N	NAPHTHALENE	150	1,600,000	19																
N	PHENANTHRENE	23,000,000	23,000,000	19																
N	PYRENE	480,000	23,000,000	19																
Pesticides (g/g)																				
C	4,4'-DDD	11,000	2,700	0.47																
C	4,4'-DDE	35,000	1,900	0.48																
N	ENDOSULFAN II	20,000	470,000	0.4																
Metals (mg/kg)																				
C	CALCIUM			62																
C	MAGNESIUM			10																
C	POTASSIUM			23.7																
C	SODIUM			10.7																
C	ARSENIC	0.026	0.43	0.09																
N	ALUMINUM	78,000	2																	
N	ANTIMONY	13	31	0.24																
N	BARIUM	2,100	5,500	0.93																
N	BERYLLIUM	127	160	0.01																
N	CADMIUM	27	39	0.02																
N	CHROMIUM (as Cr+6)	42	230	0.09																
N	COBALT			0.3																
N	COPPER	11,000	4,700	0.3																
N	IRON	23,000	3,100	0.08																
N	LEAD (EPA action level)	400	23,000	0.09																
N	MANGANESE	950	1,600	0.28																
N	MERCURY (as HgCl2)			0.05																
N	NICKEL	1,600	0.27																	
N	SELENIUM	19	390	0.14																
N	SILVER	31	390	2.2																
N	THALLIUM	3.6	5.5	0.11																
N	VANADIUM	5,100	550	0.76																
N	ZINC	14,000	23,000	0.29																
Inorganic (mg/kg)																				
C	CYANIDE	150	1,600	0.23																

COPC: Shaded and bold type: Analyte was detected above screening benchmark, or detected and benchmark not available.

CN: Carcinogenic or noncarcinogenic analyte. Full value RBC shown for all analytes. "N" analyte compared to 1/10 of RBC. Selected metal species for RBC indicated.

B: Not detected substantially above level in laboratory or field blank.

J: Estimated value or quantitation limit.

W: Post digestion spike is out of control limits and sample absorbance is less than 50% of spike absorbance.

BLANK CELL INDICATES ANALYTE NOT DETECTED.
U: Not detected
R: Unusable result
• Duplicate analysis is not within control limits
L: Value or quantitation limit may be higher

TABLE 4-5 GROUNDWATER ANALYTE DETECTIONS AND COPCS, PI-1

Sample Location: Depth(ft): Sample Date:					MW1	MW1-DUP	MW2
					26	26	24.5
					10/26/1999	10/26/1999	10/26/1999
C/N	Analytes	MCLs	Tap Water RBC	Reporting Limit			
	Volatiles (ug/L)						
C	1,1,2,2-TETRACHLOROETHANE		0.053	0.2	0.2 U	0.2 U	0.2 U
C	1,1,2-TRICHLOROETHANE	5	0.19	0.4	0.4 U	0.4 U	0.4 U
C	1,1-DICHLOROETHYLENE	7	0.044	0.6	0.6 U	0.6 U	0.6 U
C	1,2-DICHLOROETHANE	5	0.12	0.4	0.4 U	0.4 U	0.4 U
C	1,2-DICHLOROPROPANE	5	0.16	0.4	0.4 U	0.4 U	0.4 U
C	BROMODICHLOROMETHANE	100	0.17	0.4	0.4 U	0.4 U	0.4 U
C	CARBON TETRACHLORIDE	5	0.16	0.4	0.4 U	0.4 U	0.4 U
C	CHLORODIBROMOMETHANE	100	0.13	0.2	0.2 U	0.2 U	0.2 U
C	CHLOROFORM	100	0.15	0.2	0.2 U	0.2 U	0.2 U
C	VINYL CHLORIDE	2	0.019	0.6	0.6 U	0.6 U	0.6 U
	Semi-Volatiles(ug/L)						
C	1,4-DICHLOROBENZENE	75	0.47	2	2 U	2 U	2 U
N	1,3-DICHLOROBENZENE		5.5	2	2 U	2 U	2 U
C	2,2'-OXYBIS(1-CHLOROPROPANE)		0.26	1	1 U	1 U	1 U
C	3,3'-DICHLOROBENZIDINE		0.15	7	7 U	7 U	7 U
C	BIS(2-CHLOROETHYL) ETHER		0.0096	2	2 U	2 U	2 U
C	BIS(2-ETHYLHEXYL) PHTHALATE	6	4.8	2			1.6 J
C	HEXACHLORO-1,3-BUTADIENE		0.86	2	2 U	2 U	2 U
C	HEXACHLOROBENZENE	1	0.042	3	3 U	3 U	3 U
C	N-NITROSODI-N-PROPYLAMINE		0.0096	4	4 U	4 U	4 U
C	PENTACHLOROPHENOL	1	0.56	2	2 U	2 U	2 U
N	2,4-DINITROPHENOL		73	23	23 U	23 U	23 U
N	2-METHYL-4,6-DINITROPHENOL		3.7	5	5 U	5 U	5 U
N	NITROBENZENE		3.5	3	3 U	3 U	3 U
	PAHs (ug/L)						
C	BENZO[A]PYRENE	0.2	0.0092	0.05	0.05 U	0.05 U	0.05 U
C	DIBENZ[A,H]ANTHRACENE		0.0092	0.05	0.05 U	0.05 U	0.05 U
	Pesticides (ug/L)						
N	ENDOSULFAN SULFATE		220	0.0058	0.05 J	0.03 J	0.26
N	ENDRIN KETONE		11	0.0028	0.01 J	0.0059 J	0.03
C	4,4'-DDD		0.28	0.0036		0.01 J	
C	4,4'-DDE		0.2	0.0048	0.01 J	0.01 J	0.04
C	4,4'-DDT		0.2	0.004	0.01 J	0.02 J	
C	ALDRIN		0.0039	0.0046	0.0073 J	0.0057 J	0.07
C	ALPHA-HCH		0.011	0.002	0.0024 J		0.13
C	TOXAPHENE	3	0.061	0.09	0.09 U	0.09 U	0.09 U
C	GAMMA-HCH	0.2	0.052	0.0016	0.0075 J		0.01
C	HEPTACHLOR	0.4	0.015	0.0046			0.08
C	HEPTACHLOR EPOXIDE	0.2	0.0074	0.0038	0.01	0.01 J	
	PCBs (ug/L)						
C	AROCLOR 1221	0.5	0.033	0.06	0.06 U	0.06 U	0.06 U
C	AROCLOR 1232	0.5	0.033	0.05	0.05 U	0.05 U	0.05 U
C	AROCLOR 1242	0.5	0.033	0.06	0.06 U	0.06 U	0.06 U
C	AROCLOR 1254	0.5	0.033	0.08	0.08 U	0.08 U	0.08 U
C	AROCLOR 1260	0.5	0.033	0.08	0.08 U	0.08 U	0.08 U

BLANK CELL INDICATES ANALYTE NOT DETECTED.

COPC: Shaded and bold type: Analyte was detected above screening benchmark, or detected and benchmark not available.

B: Not detected substantially above level in laboratory or field blank.

U: Not detected.

J: Estimated value of quantitation limit.

L: Value or quantitative limit may be higher.

TABLE 4-5 (continued)

Sample Location: Depth(ft): Sample Date:					MW1	MW1-DUP	MW2
					26	26	24.5
					10/26/1999	10/26/1999	10/26/1999
C/N	Analytes	MCLs	Tap Water RBC	Reporting Limit			
Metals (ug/L)							
	CALCIUM			55.6	2340	2240	1160
	CALCIUM, DISSOLVED			55.6			1300
	MAGNESIUM			95.6	620	613	1120
	MAGNESIUM, DISSOLVED			95.6			539
	POTASSIUM			174	1130	1090	2130
	POTASSIUM, DISSOLVED			174			1270
	SODIUM			62.3	3730	3710	3740 J
	SODIUM, DISSOLVED			62.3			5740 J
C	ARSENIC	50	0.045	1.7	1.7 U	1.7 U	1.7 U
C	ARSENIC, DISSOLVED	50	0.045	1.7			1.7 U
N	ALUMINUM		37,000	37.9	80.8	59.6	7140
N	ALUMINUM, DISSOLVED		37,000	37.9			37.9 U
N	ANTIMONY	6	15	1	1 UL	1 UL	1 UL
N	ANTIMONY, DISSOLVED	6	15	1			1 UL
N	BARIUM	2000	2,600	4.5	20.7	19.9	73.5 J
N	BARIUM, DISSOLVED	2000	2,600	4.5			202 J
N	BERYLLIUM	4	73	0.2	0.3 B	0.3 B	0.2 UJ
N	BERYLLIUM, DISSOLVED	4	73	0.2			0.3 B
N	CADMIUM	5	18	0.2	0.2 U	0.2 U	0.2 U
N	CADMIUM, DISSOLVED	5	18	0.2			0.2 U
N	CHROMIUM (as Cr+6)	100	110	0.7	0.7 UL	0.7 UL	22
N	CHROMIUM, DISSOLVED	100	110	0.7			0.7 UL
N	COBALT		2,200	7.5	7.5 UL	7.5 UL	14.2 L
N	COBALT, DISSOLVED		2,200	7.5			8.9 L
N	COPPER	1300	1,500	2.1	2.1 UL	2.1 UL	2.1 UL
N	COPPER, DISSOLVED	1300	1,500	2.1			2.1 UL
N	IRON		11,000	5.5	59.9 B	62 B	5390
N	IRON, DISSOLVED		11,000	5.5			24.4 B
N	LEAD (EPA action level)	15	15	1.1	1.1 U	1.1 U	13.8
N	LEAD, DISSOLVED	15	15	1.1			1.1
N	MANGANESE		730	2.4	49.5	46.9	495
N	MANGANESE, DISSOLVED		730	2.4			395
N	MERCURY (as HgCl2)	2	11	0.1	0.1 U	0.1 U	0.1 U
N	MERCURY, DISSOLVED	2	11	0.1			0.1 U
N	NICKEL	100	730	13.1	13.1 U	13.1 U	20.8
N	NICKEL, DISSOLVED	100	730	13.1			13.1 U
N	SELENIUM	50	180	1.8	1.8 U	1.8 U	1.8 U
N	SELENIUM, DISSOLVED	50	180	1.8			1.8 U
N	SILVER		180	2.2	2.2 U	2.2 U	2.2 U
N	SILVER, DISSOLVED		180	2.2			2.2 U
N	THALLIUM	2	2.6	1	1 U	1 U	1 U
N	THALLIUM, DISSOLVED	2	2.6	1			1 U
N	VANADIUM		260	5.6	5.6 U	5.6 U	8.9
N	VANADIUM, DISSOLVED		260	5.6			5.6 U
N	ZINC		11,000	1.7	1.7 UL	1.7 UL	16.1 J
N	ZINC, DISSOLVED		11,000	1.7			65.6 J

BLANK CELL INDICATES ANALYTE NOT DETECTED. MW-1 and DUP did not include dissolved analysis.

COPC: Shaded and bold type: Analyte was detected above screening benchmark, or detected and benchmark not available. Selected metal species for RBC indicated.

B: Not detected substantially above level in laboratory or field blank.

U: Not detected.

J: Estimated value of quantitation limit.

L: Value or quantitative limit may be higher.

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TABLE 4-6 GROUNDWATER SAMPLING FIELD QA/QC BLANK RESULTS,
PI-1

Field Sample ID:	MW-EB1	PII-MW-FB	TRIP BLANK GW-2
Sample Type:	RB	FB	TB
Associated Matrix:	GW	GW	GW
Sample Date:	10/26/1999	10/26/1999	10/26/1999
Inorganic (mg/L)			
CYANIDE	0.01 UL	0.01 UL	
Metals (ug/L)			
ALUMINUM	37.9 UJ	37.9 U	
ALUMINUM, DISSOLVED	38.5 J		
ANTIMONY	1 UL	1 UL	
ANTIMONY, DISSOLVED	1 UL		
ARSENIC	1.7 U	1.7 U	
ARSENIC, DISSOLVED	1.7 U		
BARIUM	4.5 UL	4.5 UL	
BARIUM, DISSOLVED	4.5 UL		
BERYLLIUM	0.3 B	0.3 B	
BERYLLIUM, DISSOLVED	0.3 B		
CADMIUM	0.2 U	0.2 U	
CADMIUM, DISSOLVED	0.2 U		
CALCIUM	102 B	109 B	
CALCIUM, DISSOLVED	109 B		
CHROMIUM	0.7 UL	0.7 UL	
CHROMIUM, DISSOLVED	0.7 UL		
COBALT	7.5 UL	7.5 UL	
COBALT, DISSOLVED	7.5 UL		
COPPER	2.1 UL	2.1 UL	
COPPER, DISSOLVED	2.1 UL		
IRON	5.5 UJ	18.1 B	
IRON, DISSOLVED	37.6 B		
LEAD	1.1 U	1.1 U	
LEAD, DISSOLVED	1.1 U		
MAGNESIUM	95.6 U	95.6 U	
MAGNESIUM, DISSOLVED	95.6 U		
MANGANESE	2.4 UL	2.4 UL	
MANGANESE, DISSOLVED	2.4 UL		
MERCURY	0.1 UJ	0.1 U	
MERCURY, DISSOLVED	0.14 B		
NICKEL	13.1 U	13.1 U	
NICKEL, DISSOLVED	13.1 U		
POTASSIUM	174 UL	174 UL	
POTASSIUM, DISSOLVED	174 UL		
SELENIUM	2.3 B	1.8 U	
SELENIUM, DISSOLVED	1.8 U		
SILVER	2.2 U	2.2 U	
SILVER, DISSOLVED	2.2 U		
SODIUM	504 B	530 B	
SODIUM, DISSOLVED	554 B		
THALLIUM	1 U	1 U	
THALLIUM, DISSOLVED	1 U		
VANADIUM	5.6 U	5.6 U	
VANADIUM, DISSOLVED	5.6 U		
ZINC	1.7 UL	1.7 UL	
ZINC, DISSOLVED	1.7 UL		
SVOC(ug/L)			
BIS(2-ETHYLHEXYL) PHTHALATE	2 U	2 U	
VOA (ug/L)			
ACETONE	3 U	3 U	3 U
BROMODICHLOROMETHANE	6 B	6 B	0.4 U
CHLOROFORM	32 B	36 B	0.2 U
METHYLENE CHLORIDE	4 B	5 B	0.5 U
TOLUENE	0.5 U	0.5 U	0.5 U

B: Not detected substantially above level in laboratory or field blank.

J: Estimated value of quantitation limit.

U: Not detected.

TB: Trip blank.

GW: Groundwater

RB: Rinse blank.

FB: Field blank.

L: Value or quantitative limit may be higher.

TABLE 4-7 GROUNDWATER QUALITY PARAMETERS, PI-1, 26 OCTOBER 1999

Sample	pH	Conductivity (mS/cm)	Temperature (C)	Dissolved Oxygen (mg/L)	Oxidation/ Reduction (mV)	Turbidity (NTU)
MW-1	4.36	26	20.2	3.64	256.6	8.4
MW-2	4.79	36	25.99	4.87	169	157

TABLE 4-8 SUMMARY OF COPCs BY MEDIA, PI-1, FORT PICKETT (a)

Analyte	Onsite Surface Soil *	Subsurface Soil	Groundwater	Downslope Surface Soil *
VOCs	ug/kg			
Acetone	21-750: 8E			110: 1E
SVOCs	ug/kg			
Dibenzofuran	180: 1E			
PAHs	ug/kg			ug/kg
2-Methylnaphthalene	160-660: 2E			
Chrysene	130: 1E			
Benzo(b)fluoranthene	150: 1E			
Fluoranthene	110: 1E			
Naphthalene	270: 1E			
Phenanthrene	360: 1E			
Pyrene	270: 1 E			120: 1 E
Pesticides			ug/L	
Aldrin			0.0073-0.07: 2	
Alph-HCH			0.13: 1	
Heptachlor			0.08: 1	
Heptachlor epoxide			0.01: 1	
Dioxins/Furans	ng/kg			
TEQ	98.5: 1HH			
TAL Metals	mg/kg	mg/kg	ug/L	mg/kg
Aluminum	2,130-9,410: 8E,1HH	10,700-36,600: 14		4,850: 1E
Antimony	0.5-1.1: 3E			0.66: 1E
Arsenic	0.89-3.1: 8HH	0.61-2.2: 14		2.1: 1HH
Barium				
Beryllium	0.17-0.3: 8E			0.3: 1E
Cadmium				
Calcium				
Chromium	4.6-9.6: 8E	33.1: 1		10: 1E
Cobalt				
Copper	16-33.1: 2E			
Iron	4,910-14,200: 8HH	4,050-54,900: 15		11,500: 1HH
Lead	11.5-118: 8E			118: 1E
Magnesium				
Manganese	117-131: 2E	172-399: 6	395 :1	105: 1E
Mercury	0.06-0.1: 2E			0.06: 1E
Nickel	2.7-9.1: 5E			2.2: 1E
Potassium				
Selenium				
Silver				
Sodium				
Thallium	0.14-0.19: 4E	0.73-3.8: 6		0.23: 1E
Vanadium	5.4-18.2: 8E	99.8: 1		13.2: 1E
Zinc	24.7-181: 6E			124: 1E

(a) COPCs include detected analytes in media with concentrations above screening criteria.

(Concentration range: Number of sample locations that exceeded the screening criteria)

* For surface soil, number of exceedances according to ecological (E) or RBC (HH) benchmarks is shown.

"Downslope" soil refers to SS-9 in swale outside former maintenance area. All surface soil samples were included in risk assessment for the site.

BOLD TYPE: indicates analyte exceeds risk-based criteria (if available).

NORMAL TYPE: indicates appropriate screening criteria were not available for detected analyte.

**TABLE 4-9 ANALYTES NOT DETECTED WITH REPORTING LIMITS
ABOVE SCREENING CRITERIA AT PI-1**

Analyte	MEDIA		
	Surface Soil	Subsurface Soil	Groundwater
VOCs			
1,1,2,2-Tetrachloroethane			*
1,1,2-Trichloroethane			*
1,1-Dichloroethylene			*
1,2-Dichloroethane			*
1,2-Dichloropropane			*
Bromodichloromethane			*
Carbon tetrachloride			*
Chlorodibromomethane			*
Chloroform			*
Vinyl chloride			*
SVOCs			
1,4-Dichlorobenzene			*
1,3-Dichlorobenzene			*
2,2'-Oxybis(1-chloropropane)			*
n-Nitrosodi-n-propylamine	*	*	*
3,3'-Dichlorobenzidine			*
Bis(2-chloroethyl)ether			*
Hexachloro-1,3-butadiene			*
Hexachlorobenzene			*
Pentachlorophenol			*
2,4-Dinitrophenol			*
2-Methyl-4,6-dinitrophenol			*
Nitrobenzene			*
PAHs			
Benzo(a)pyrene			*
Dibenz(a,h)anthracene			*
Pesticides			
Toxaphene			*
PCBs			
			*
Metals			
Arsenic			*
Thallium			*

5. CONTAMINANT FATE AND TRANSPORT

5.1 OBJECTIVES

This chapter provides further assessment of Site PI-1 by evaluation of environmental fate and mobility of COPCs. Media evaluated include surface soil, subsurface soil, and groundwater. COPCs relative to RBCs, MCLs, and ecotoxicity criteria were identified in Chapter 4. This chapter provides an evaluation of persistence of the COPCs in the environment and the potential for inter-media transfer and migration offsite. Specific objectives are to:

- Describe the potential routes of COPC migration
- Estimate the persistence of COPCs in the study area based upon physical, chemical, and biological factors that affect fate and mobility
- Predict migration routes of COPCs that may allow for human exposure or may adversely impact the environment

5.2 POTENTIAL CONTAMINANT SOURCES

PI-1 was formerly a vehicle repair and maintenance shop. Given this history, it is possible that contaminants such as greases, oils, solvents, vehicle emissions, and petroleum products may have impacted the site. A concrete slab with a soil-filled maintenance pit and a soil-filled concrete void possibly used for a hydraulic lift were observed during the field effort. Based on the configuration of the slab, it appeared that vehicles were driven onto the slab and over the maintenance pit for repair. The presence of former toilet vents and drains on the southwest corner of the slab indicated the potential presence of a septic system, however, its location and construction details were unknown. A metal detector survey during the field study detected an anomaly near boring SB-3. No structure was intercepted during drilling. Aerial photos showed disturbed or stained ground in formerly used storage areas south of the concrete slab. An approximately 2.5-ft diameter shallow well was observed on the west side of the slab. This well was likely used as an onsite potable water supply. No sheen was observed on the water in the well and there were no detected organic vapors in the well. No structures were observed that indicated the possible presence of underground tanks at the site.

The site is located adjacent to the northwest side of a currently cultivated field. At the time of this field investigation, tobacco was being grown. It is possible that pesticides may have been applied to the field. Pesticide application also occurred in the past for activities at Fort Pickett.

5.3 POTENTIAL ROUTES OF MIGRATION

A route of migration is a pathway from a source or mechanism of chemical release into the surrounding area by the way of a transport medium. These routes can be natural or man-made pathways. Surface runoff, groundwater flow, and air transport were evaluated. Subsurface and surface soil, although not necessarily transport media, may be considered a source if sufficiently contaminated, which can affect these pathways.

5.3.1 Surface Runoff

Surface runoff (water and/or eroded soil) can provide a pathway for chemical constituents to migrate offsite via channelized or sheet flow. The topography of PI-1 is generally flat, with a gentle slope towards the northwest. The elevation of the site ranges from 434 to 438 ft above mean sea level. The area of PI-1 has been cleared of trees and unpaved areas are covered with grass. A drainage swale parallel to Route 460 conveys stormwater runoff westward off of the site. The soil series of the site has high available moisture capacity and poor natural drainage.

As a result of the site topography and soil profile, there is little potential for erosion of soil and formation of runoff channels by surface water runoff in any vegetation-bare areas. Based upon the presence of COPCs in surface soil (0-6 in.), there is the potential, though limited, for transport of COPCs by overland surface water flow through soil erosion or dissolution of COPCs from soil to surface runoff. This observation is supported by the presence of COPCs in surface soil samples collected at an offsite downgradient swale, though it is not known if PI-1 actually contributed to the elevated metals and pyrene in the swale. Sheet flow may be present in areas of the site. In the absence of a significant grade on the site, direct infiltration of surface water may also be a route. In addition, due to the soil composition, the potential for standing water to interact with soil and dissolve COPCs from the soil exists. This standing water could then be transported as surface water runoff or direct infiltration. In summary, surface water transport of COPCs may be considered a migration route of concern.

5.3.2 Groundwater Flow

The groundwater-flow pathway is formed by rain and snowmelt infiltration or direct water-table contact. COPCs that leach into soil pore water are then transported via groundwater flow from high to low hydraulic head. Three flow components must be considered for this pathway: horizontal flow, vertical flow, and time.

During the RI, groundwater elevations were gauged with depths to water ranging 15 to 16 ft below the ground surface. In general, groundwater flows across the site in a southeast to northwest direction (Figure 3-2). The groundwater migration pathway is a potential pathway of concern as all of the necessary components (infiltration, leaching, vertical, and advective flow) are present at the site. The processes of advection and dispersion are discussed in detail in Section 5.3.

5.3.3 Air Transport

The air transport pathway is formed by wind movement of surficial soil particles or volatile vapors venting from soil. Particle transport is limited by the particle size, wind speed, and surface conditions. The amount of volatile venting is controlled by chemical properties, soil types, and surface conditions.

Site PI-1 has a thick grass cover, and as a result, particle transport via wind is limited. Based on the mapped soil associations for the area, the surface soil is composed of a mixture of silt and sand, mostly a sandy loam. It appears that this surface-A horizon has been eroded over most of the site based on the boring logs where the typical subsoil for the area occurs at the surface. The subsoil is a friable clay loam with fine mica. This soil type has high available moisture capacity and, if it contains adequate moisture, would not be subject to wind erosion.

5.4 CONTAMINANT PERSISTENCE AND MIGRATION

5.4.1 Transport and Fate Processes

The main, physical transport processes in groundwater include the following:

- **Advection:** This term describes mass transport due simply to the flow of water in which the mass is dissolved. The direction coincides with that of groundwater and the rate of transport is proportional to the groundwater flow rate. Advection is normally the more important of the two transport processes.
- **Hydrodynamic Dispersion:** This term describes a process of fluid mixing, through molecular diffusion and mechanical dispersion, that causes a zone of mixing to be developed between a fluid of one composition that is adjacent to or being displaced by a fluid of another composition.

Other reactions that affect COPC mobility and fate include chemical and biological processes. The major categories are acid-base reactions, solution, volatilization, precipitation, complexation, sorption reactions, oxidation-reduction reactions, hydrolysis reactions, and isotopic reactions.

Sorption is an important reaction in retarding migration of organic compounds. Sorption of organic compounds to soil or sediment is quantified by its distribution (sorption) coefficient, K_d ,

$$K_d = \frac{C_s}{C_e} \quad (\text{Eq. 1})$$

which is defined as:

where:

- C_s = concentration of a contaminant sorbed to a specific weight of sediment
- C_e = concentration of the same contaminant dissolved in an equal weight of water
(Olsen et al. 1982)

The sorption coefficient is commonly expressed on an organic-carbon basis and is defined as:

$$K_{oc} = \frac{K_d}{f_{oc}} \quad (\text{Eq. 2})$$

where:

- f_{oc} = fractional mass of organic carbon in the sediment/soil

The soil-water partition coefficient K_{oc} can be estimated from either the water solubility or the octanol-water partition coefficient (K_{ow}). This value provides an indication of the tendency of a chemical to partition between particles containing organic carbon content and water. The greater the K_{oc} , the less likely the constituent will partition from soil to water. The octanol-water partition coefficient has been recognized as the key parameter in predicting the environmental fate of organic compounds (Montgomery and Welkom 1990). An organic compound's octanol-water partition coefficient, K_{ow} , is defined as the ratio of the compound's concentration in a known volume of n-octanol to its concentration in a known volume of water after the octanol and water have reached equilibrium (Smith et al. 1988).

One of the major factors controlling the migration of organic compounds in porous media is the organic carbon content of the porous media. The distribution coefficient (K_d) for a given solute is directly proportional to the organic content of a soil or sediment (equation 2); the larger the organic carbon content, the greater the value of K_d (Karickhoff et al. 1979). This approach in estimating K_d is applicable to soil typically containing organic matter greater than 0.001 (0.1 percent). For soil that contains lower organic carbon content, sorption of neutral organics onto the mineral phase can cause erroneous K_d estimates (Chiou et al. 1985).

A value of six percent was used as an estimate for TOC in surface soil based on a TOC of 67,300 mg/kg in SB8-0.5. For subsurface soil, TOC was not detected above 3,760 mg/kg. A value of one percent was estimated for subsurface soil.

Organic compounds present in the environment will eventually reduce to elemental forms (e.g., carbon, hydrogen, and oxygen) with exposure to water, air, bacteria, soil, sunlight, or combinations thereof. Organic compound persistence is of concern because of their ability to bioaccumulate in ecological receptors. Persistence of organic COPCs in the environment is related to resistance to degradation. Using published data, the organic COPCs in soil and groundwater were examined to assess the rates of degradation.

The rate of degradation is presented as a half-life, which represents the amount of time necessary to reduce half the quantity of the constituent. Half-lives were based upon the high and low degradation rates of the most important degradation process within a particular medium. Additional source loading is not included in degradation values presented in this report.

Compounds in soil are degraded predominantly by biodegradation, with the exception of those compounds that undergo hydrolysis. Soil and water hydrolysis half-lives were used interchangeably, even though higher organic matter concentrations may increase the hydrolysis rate for acid-catalyzed reactions or slow the hydrolysis rate for base-catalyzed reactions (Perdue 1983). Most hydrolysis half-lives in soil were based upon rate data determined in water, since actual hydrolysis data in soil were rarely found (Howard et al. 1991).

Compounds in groundwater are degraded principally by biodegradation and, to a lesser extent, hydrolysis. Grab sample or field studies with dependable rate data are seldom found in the literature for compounds in groundwater. Generally, biodegradation in groundwater proceeds at a slower rate compared to surface water, because groundwater microbial populations may be limited in numbers and enzymatic capabilities based on sufficient recharge of electron acceptors

and nutrients and a slow response to changing plume chemistry. Groundwater can maintain varying levels of oxygen and are more likely (than surface water) to be anaerobic (Howard et al. 1991), which further reduces microbial populations.

Therefore, the rate of biodegradation in groundwater was assumed to be one-half that in surface water, and overall half-lives were conservatively estimated to be twice the unacclimated aqueous aerobic biodegradation half-lives, unless the data suggested otherwise. For compounds that degrade rapidly under anaerobic conditions, the low half-life in groundwater usually equaled to the low aqueous anaerobic biodegradation half-life, and the high half-life was based upon the high aqueous aerobic biodegradation half-life (Howard et al. 1991).

The half-lives presented in this report are based on estimated times of degradation from readily available data. For soil half-lives, dissolution into infiltrating groundwater was not considered. For groundwater half-lives, physical transport parameters such as advection and diffusion were not considered. For both soil and groundwater, the concentration was considered unaffected by additional source loading.

Degradation rates can be used to estimate the length of time necessary to lower a compound concentration. Soil half-lives (with limitations described above) provide an estimate for natural *in situ* attenuation of the compound.

5.4.2 Volatile Organic Compounds

The primary fate process for VOCs with exposure to the atmosphere is volatilization. However, in soil and groundwater environments, secondary processes occur, such as biological and chemical degradation. Table 5-1 lists the half-life range, sorption coefficient, solubility, and vapor pressure for the organic COPCs identified at PI-1. Half-lives are presented for compounds in media where they were identified as COPC.

Acetone was the only VOC COPC in surface soil at PI-1. Acetone is miscible in water and has a high vapor pressure (Table 5-1). Acetone tends not absorb to soil and, considering its chemical properties, would readily dissolve into water and evaporate from the surface soil. The short half-life of acetone is reflective of its tendency to quickly degrade in the atmosphere and to biodegrade easily (Howard 1990).

5.4.3 Semivolatile Organic Compounds, PAHs, and Dioxin/Furans

Dibenzofuran was a SVOC COPC in surface soil. Dibenzofuran is used as a research chemical. It is derived from coal gasification processes (Montgomery and Welkom 1990). Its occurrence at the site is likely residual from vehicle emissions during past vehicle repair operations. As with dibenzofuran, the occurrences of the PAHs and the dioxin/furan compounds at the site are likely residual compounds from vehicle emissions.

The higher molecular weight PAHs and dioxin/furan compounds are typically persistent compounds due to low solubilities and high partition coefficients, where the compounds will strongly partition from water to particulate and dissolved organic matter. PAH half-lives can be extended by sorption to organic carbon. Gardner et al. (1979) provided evidence that PAH slowly biodegrade after being sorbed onto sediment. Half-lives will typically be much longer for PAHs than VOCs due to increased complexity of the compounds.

Conversely, the solubilities for dibenzofuran, naphthalene, and 2-methylnaphthalene are higher. These compounds may dissolve appreciably in surface or groundwater. Thus, these compounds are more mobile than high molecular-weight PAHs, but not as mobile as VOCs such as benzene.

Available solubilities and sorption coefficients (K_{oc}) for the COPC compounds are listed in Table 5-1. Consequently, the ultimate fate of these compounds is most likely sorption to soil, followed by slow biodegradation at half-life rates listed in Table 5-1. Naphthalene and dibenzofuran compounds may tend to migrate with surface or groundwater. Values for a dioxin and a furan congener are presented for typical characteristics of dioxin/furan compounds. Other transport and transformation processes, such as volatilization and hydrolysis, are not important fate processes for these compounds (Smith et al. 1988).

5.4.4 Pesticides

Four pesticides were identified as COPC in groundwater. Aldrin, alpha-HCH, and gamma-HCH were detected at MW1 and MW2. Aldrin was used as an insecticide and fumigant and its use was stopped in 1987 in the United States (Briggs 1992). Aldrin transforms into dieldrin over time but dieldrin was undetectable in both groundwater samples. Alpha- and gamma-HCH are the alpha and gamma isomers, respectively, of benzene hexachloride. Gamma-HCH, also known as lindane, was used as insecticide until use was cancelled in 1983 (Briggs 1992).

Heptachlor was detected in MW2. Heptachlor was used as an insecticide for termite and ant control. It was banned from use in 1984 in the United States, but is still allowed for use in power transformers for fire ant control (Briggs 1992). Heptachlor transforms by oxidation to heptachlor epoxide, which is also a COPC in groundwater. Heptachlor epoxide was detected in MW1.

Fate characteristics of these compounds are included in Table 5-1. Considering relative mobility potential increases as the K_d (or K_{oc}) decreases, aldrin is the most mobile, with a K_d value of 4.07, followed by alpha-HCH (K_d of 19), heptachlor epoxide (K_d of 210) and the least mobile, heptachlor (K_d of 220). All four of these pesticides contain low vapor pressures and low solubilities, relatively high octanol-water partition coefficients, high soil-water partition coefficients, and high soil absorption coefficients. These compounds strongly partition from water into particulate and organic matter, therefore, they are not expected to migrate long distances with water in dissolved form. Consequently, the ultimate fate is limited sorption to soil, followed by slow biodegradation. However, volatilization from groundwater or surface water to air will not be a significant pathway due to low vapor pressures (Briggs 1992).

These data indicate the strong tendencies of pesticides with low solubilities to partition into organic matter (Smith et al. 1988). Half lives are typically much longer for organochlorine pesticides than VOCs due to increase chemical complexity of the compounds. Pesticide half lives can also be extended by sorption to organic carbon.

5.4.5 Inorganics

Several metal COPCs were identified in soil. Manganese was a COPC in groundwater. The potential for transport of metals in the subsurface is based upon their combined affinity for either soil or water. Factors controlling the process include the soil-water chemistry and the charge deficiency on adsorbent surfaces (i.e., soil or sediment). An accumulation of ions near the solid/liquid interface is required to neutralize this surface charge; the clay mineral fraction is most commonly involved, for example, a soil or sediment charge deficiency can be neutralized by exchange of ions in solution. Specific factors such as soil texture, soil chemistry, pH, redox potential, and solute and ligand concentrations in pore water further control the mobility of a particular metal species. In general, the solubility of metals tends to increase with increasing acidity, and conversely, lower mobility under neutral and slightly alkaline conditions.

A number of mechanisms and processes are involved in metal attenuation (or sorption), which generally refers to the removal of solute components from the aqueous phase onto the solid

phase surfaces. These attenuation processes can be classified as either physical or chemical. Physical adsorption is due to surface charges that then attract ionic species of the opposite charge. Isomorphic substitution is the principal cause of surface charges. For example, clay minerals tend to have a negative surface charge and therefore tend to attract positively charged ions. This process is reversible in some instances due to the low energy of adsorption.

Hydrous oxides, particularly of iron and manganese, may also be involved in the sorption of metals. A high degree of disorder characterizes freshly precipitated hydrous oxides, which are frequently gelatinous with large surface areas and reactivities. Metal ions are sorbed onto these surfaces and essentially become co-precipitated or occluded with the hydrous oxides. Metal oxides are of further importance because they often occur as coatings on other surfaces.

Chemical processes for metals include ion exchange, organic complexation, precipitation and co-precipitation, solid-state diffusion, and isomorphic substitution. Organic compounds can be formed with some metals in soil, biotically or abiotically. Metals may also sorb to sediment via interaction with the organic matter or clay particles to form stable complexes that are relatively insoluble in water.

Groundwater transport of metals can also involve transport by colloids, although no site-specific data have been collected to assess this pathway. Although there is disagreement as to the exact particle dimensions involved in colloid transport, colloids are generally believed to range from 0.001 to 1 μm . The small size of these particles may allow them to remain suspended in solution by Brownian motion. If colloid particles are mobile over long distances, then they can potentially facilitate transport through processes such as adsorption onto mobile surfaces or by precipitation reactions with the surface. Colloids can originate from a number of sources, including detachment of particles from immobile soil and mineral matrices, supersaturated mineral solutes such as iron oxides, viruses and small bacteria, emulsions or microemulsions of fine droplets, macromolecules such as agglomerated humic acid molecules, and colloids directly introduced by areal sources such as landfill leachate (Mills et al. 1991).

To be mobile over long distances, suspended colloidal material must be stable (resisting aggregation with other like particles) and must not be susceptible to particle filtration when passing through a porous medium. Filtration can occur either by straining, which is dependent on the pore dimensions and geometry relative to particle size, or by physical-chemical processes that are primarily controlled by electrostatic, chemical, or van der Waal's forces. It appears that several potentially important conditions must be satisfied for colloidal transport to be significant.

The media must be porous or fractured and flow rates must be relatively high (McCarthy and Zachara 1989). No site-specific data have been collected to assess this pathway.

Factors that influence metal attenuation in the subsurface are summarized in Table 5-2. In general, as with VOCs, the distribution or absorption coefficient of a metal is an indicator of the relative affinity of a metal to solids and, therefore, its immobility. Distribution coefficients for selected metals estimated from laboratory studies are presented in Table 5-3. Considering that the groundwater pH is about 4-5, available values for pH of 4.9 are shown. The manganese COPC in groundwater may be due to these acidic conditions as this metal is dissolved from the surrounding soil, and is transported in its more reduced and mobile divalent form. This condition may be natural for the area and not due to a site-specific release.

5.5 SUMMARY

The basic physiochemical principles that govern fate and mobility of chemicals were applied to site analytical data to highlight those COPCs that have a potential to be persistent or mobile along the exposure pathways identified at the site. Three pathways were evaluated: surface runoff, groundwater flow, and air transport. According to site topography and COPCs that were found in surface soil, there is the potential for offsite transport. The transport of metals via the surface water pathway may be a transport concern. Since metals can adhere to soil via physical adsorption, oxidation to hydrous oxide compounds, and formation of insoluble organic complexes, transport of adhered metal COPCs may occur during prolonged surface runoff events. Conceptually, the groundwater migration route was identified as an exposure route where offsite exposure to groundwater would be possible, although based on the surrounding land use and absence of nearby wells, this is highly unlikely. However, the potential for contaminants in groundwater to migrate offsite and the potential for risk can not be discounted. Air transport was not considered a significant pathway considering vegetation and soil conditions.

TABLE 5-1 ESTIMATED PROPERTIES FOR ORGANIC COPC^(a)

Compound	Media	Half-life Range	K _{oc} (L/Kg)	Log K _{ow}	Solubility (mg/L)	Vapor Pressure (mm Hg)
Acetone	Soil/water	1-7 days	NA	2.49	Miscible	180 at 20°C
Dibenzofuran	Soil/water	1-4 weeks	1.0 x 10 ⁴	4.2	10 at 25°C	NA
2 Methylanthralene	Surface water	2.3-440 days	7.9 x 10 ³	3.9	25 at 25°C	NA
Naphthalene	Soil/water	20-48 days	1.0 x 10 ³	3.3	30 at 25°C	0.2 at 25°C
Chrysene	Soil	1-2.7 years	2.5 x 10 ⁵	5.6	0.006 at 25°C	6.3x10 ⁻⁷ at 20°C
Phenanthrene	Soil	16-200 days	2.3 x 10 ⁴	4.5	1.18 at 25°C	6.8x10 ⁻⁴ at 25°C
Pyrene	Soil	0.5-5.1 years	6.3 x 10 ⁴	5.0	0.13 at 25°C	6.9x10 ⁻⁷ at 25°C
Fluoranthene	Soil	140-440 days	4.2 x 10 ⁴	5.2	0.26 at 25°C	5.0x10 ⁻⁶ at 25°C
Benzo(b)fluoranthene	Soil	1-1.7 years	5.5 x 10 ⁵	6.6	0.0012 at 25°C	5.0x10 ⁻⁷ at 25°C
1,2,3,4,7,8- Hexachlorodibenzo-p-dioxin	Surface water	77 days (average)	1.0 x 10 ⁶	10.4	4.4 x 10 ⁻⁶ at 20°C	2.4x10 ⁻⁶ at 25°C
2,3,4,7,8-Pentachlorodibenzo- furan	NA	NA	3.9 x 10 ⁵	6.9	5.2 x 10 ⁻⁴ at 25°C	2.6x10 ⁻⁶ at 25°C
Aldrin	Groundwater	1 day – 3.2 years	407	5.5	0.017 at 25°C	2.3x10 ⁻⁵ at 20°C
Alpha-BHC	Groundwater	13-270 days	1.9 x 10 ³	3.7	2.0 at 25°C	2.5x10 ⁻⁵ at 20°C
Heptachlor	Groundwater	1-5 days	2.2 x 10 ⁴	5.4	0.056 at 25°C	4.0x10 ⁻⁴ at 25°C
Heptachlor epoxide	Groundwater	1-1,101 days	2.1 x 10 ⁴	4.0	0.3 at 25°C	2.6x10 ⁻⁶ at 20°C

(a) References: Montgomery and Welkom (1990), Howard et al. (1991), Mackay et al. (1992)

NA: Data not available.

TABLE 5-2 PHYSICAL AND CHEMICAL PROPERTIES OF SOIL, SOIL SOLUTION,
AND SOLUTE THAT AFFECT ATTENUATION

Soil Solids	Soil Solution	Solute
Composition: Hydrous oxides (Fe, Mn, Al) Silicates clay content, type of clay Organic material Carbonate minerals Specific area Cation exchange capacity (CEC) pH (content of basic species) Eh [content of redox active species, e.g., Fe(II), Mn(IV)] Aeration status (saturated, unsaturated) Microbial type and population Temperature	pH Eh Temperature Ionic strength Ionic composition: Competing ions Complexing ions	Chemical identity Chemical behavior: Charge Size Complexation chemistry Solubility Precipitation chemistry Redox chemistry (oxidation states)

Source: Bodek et al. 1988

TABLE 5-3 SOIL-WATER DISTRIBUTION COEFFICIENTS (K_d) FOR INORGANIC CONSTITUENTS OF POTENTIAL CONCERN

Parameter	$K_d(\text{pH} = 4.9)^{(a)}$	Range of K_d (L/kg) ^(b)
Aluminum	NA	NA
Antimony	45 est.	2 – 550
Arsenic (+3)	25	1 – 19
Beryllium	23	70 – 8,000
Chromium (+3)	1.2×10^3	168 – 3,600
Copper	NA	NA
Lead	NA	NA
Manganese	NA	NA
Mercury	0.04	10 – 5,280
Nickel	16	12 – 4,700
Thallium	44	0 – 1,500
Vanadium	1,000 est.	
Zinc	16	0.1 – 100,000

(a) MINTEQ model results (USEPA 1996b).

(b) Summary of literature values (USEPA 1996b).

NA=data not available

6. HUMAN HEALTH RISK ASSESSMENT

6.1 INTRODUCTION

The purpose of this human health risk assessment (HHRA) is to determine whether there are potential human health risks associated with COPCs in environmental media at Site PI-1. The HHRA for PI-1 was conducted in accordance with U.S. EPA Risk Assessment Guidance for Superfund (RAGS) Part A (EPA 1989b) and with Region III technical guidance for risk assessment manuals (EPA 1993a and 1995b).

The risk assessment methodology used in this HHRA involves a four-step process: hazard identification, exposure assessment, toxicity assessment, and risk characterization. A brief description of each step is provided below:

- In the *hazard identification*, environmental monitoring data are evaluated, COPCs are selected for inclusion throughout the remainder of the risk assessment, and the rationale for their selection is documented.
- In the *exposure assessment*, the human population, or groups of individuals potentially exposed to COPCs, (i.e., potential human receptors) are characterized. From the many potential pathways of exposure, pathways applicable to potential receptors at the site are identified. The concentrations of COPCs in relevant media (e.g., surface soil, air) are converted into systemic doses, taking into account rates of contact (e.g., ingestion rates) and absorption rates of different COPCs. The magnitude, frequency, and duration of these exposures are then integrated to obtain estimates of daily doses over a specified period of time (e.g., lifetime, activity-specific duration).
- In the *toxicity assessment*, the relationship between extent of exposure and extent of toxic injury or disease is estimated for each COPC. Chemical-specific toxicity values, such as cancer slope factors (SFs) and reference doses (RfDs) or reference concentrations (RfCs) for non-carcinogens are presented along with a discussion of their scientific basis and derivation.
- *Risk characterization* integrates the results of the toxicity assessment and the exposure assessment to derive quantitative estimates of human health risk, including

both the risks of cancer and of non-carcinogenic effects. The major uncertainties and limitations associated with the estimates of risk and their potential ramifications are presented in this section.

6.2 HAZARD IDENTIFICATION

A hazard identification is conducted to determine which constituents are of potential concern at the site. In the hazard identification, site-specific data are analyzed and compared to risk-based screening values. Screening is conducted for both residential and industrial exposure-based scenarios.

6.2.1 Conceptual Site Model and Identification of Potential Exposure Pathways

A conceptual site model was developed for Site PI-1 to depict the potential pathways of concern at the site and is provided in Figure 6-1.

6.2.1.1 Media of Concern

Media of concern include surface soil, subsurface soil, total soil (combined surface and subsurface soil), groundwater, and air as environmental transport media for the release of chemicals present at Site PI-1. Surface water and sediment are not present at the site.

6.2.1.2 Exposure Pathways and Receptors of Concern

An exposure pathway describes a mechanism by which a population or individual may be exposed to chemicals present at a site. A completed exposure pathway requires the following four components:

- A source and mechanism of chemical release to the environment;
- An environmental transport medium for the released chemical;
- A point of potential human contact with the contaminated medium; and
- A human exposure route at the point of exposure.

All four components must exist for an exposure pathway to be complete and for exposure to occur. Incomplete exposure pathways do not result in actual human exposure and are not included in the exposure assessment and resulting risk characterization.

The site is not currently in use. The nearby areas are used for recreational and agricultural purposes. It is possible for trespassers to access the site, and adult and adolescent trespassers are the most likely to be independent enough to have access to the site on a recreational basis. Therefore, adult and adolescent trespassers are evaluated for incidental ingestion of and dermal contact with surface soil, and for inhalation of particulate entrained in air from surface soil. Although it is unlikely that trespassers will contact deeper soils, potential exposure to volatile organic constituents (VOCs) of concern in subsurface soil is also evaluated. Surface water and sediment are not present at the site, and are, therefore, not evaluated.

Through a lease with the U.S. Army, the site (PI-1) and the surrounding property has been utilized by the Southern Piedmont Agricultural Research and Extension Center (SPAREC) since the mid 1970's. SPAREC utilized the property as a multi-purpose agricultural station focusing primarily on crop research. PI-1 is located on the northwest perimeter of the property, adjacent to fields where crop research is conducted. A concrete pad (all that remains of the old service station) has been used for hay storage.

Currently, all of the adjacent property surrounding PI-1 (1,187 acres) is in the process of being permanently transferred to the Virginia Department of Education and will continue to be utilized by SPAREC. Once PI-1 has been certified as environmentally suitable for transfer, it will be transferred to SPAREC and utilized to support their research mission.

Site PI-1 consists of 2.19 acres. Approximately half of the property has been utilized in the past to grow crops as part of the research activities. Also, there is a concrete pad (as mentioned above) that has been utilized for the storage of hay. The remaining areas of PI-1 are forested or are mowed occasionally; however, they are not utilized for research activities. Interviews with the SPAREC managers indicate that their plans are to continue using this property for the same purpose into the foreseeable future. There are no plans to utilize any of this property for residential purposes (Fort Pickett BRAC 2001).

As a conservative measure, residential adult and child, construction worker, and commercial worker exposures are also evaluated for the site. The residential scenario evaluates both an adult and child resident to account for variation in exposure parameters such as body weight in the two populations. This is a conservative measure to ensure that potential future resident children are adequately assessed for risks to human health. Carcinogenic risks are evaluated on a lifetime

risk basis; therefore, for carcinogenic risks, the two populations are combined to account for potential lifetime residential exposure to the site.

Future resident adults and children are evaluated for potential risks associated with incidental ingestion of, dermal contact with, and inhalation of particulate entrained from total soils. Assessing residential risks for total soils, which assumes contact with surface and subsurface soils, is a conservative estimate to ensure that all site soils are evaluated for any potential future use. Should volatile constituents of concern be detected in subsurface soils, the potential for inhalation of volatiles is also addressed for the future resident.

Future construction workers are evaluated for incidental ingestion of, dermal contact with, and inhalation of particulate from total soil during excavation activities. It is assumed that construction workers would contact both surface and subsurface soils; therefore exposure to total soil is evaluated. Should volatile constituents of concern be detected in subsurface soils, the potential for inhalation of volatiles is also addressed for the future construction worker.

Future commercial workers are evaluated for incidental ingestion of, dermal contact with, and inhalation of particulates from surface soil. Commercial workers typically are not involved in digging scenarios where exposure to subsurface soil or groundwater would occur. In the unlikely event that these exposures were to occur, the evaluation of the residential and construction worker scenarios for these media provides a conservative estimate of potential risks to these media. Should volatile constituents of concern be detected in subsurface soils, the potential for inhalation of volatiles is also addressed for the future commercial worker.

The aquifer under PI-1 is not currently used as a source of potable water; however, future residential use of groundwater at the site is evaluated as a conservative measure. Future residential adult and child exposure to groundwater via tap water through ingestion and dermal (showering/bathing) pathways are assessed. Inhalation is only a probable pathway of concern for groundwater when there are VOCs of concern in groundwater. Should volatile constituents be determined to be of concern at the site in groundwater via the screening assessment, inhalation of VOCs while showering will be assessed for the adult resident via the U.S. EPA Region III recommended shower model by Foster and Chrostowski (1987).

The following potential human exposure pathways were identified by media for evaluation for Site PI-1 and are listed in Figure 6-1 and Table 6-1:

Exposures via Surface Soil

Current trespassers and future commercial workers are expected to have exposures to surface soils at the site.

Current Trespasser (adult, adolescent):	Incidental ingestion of surface soils Dermal contact with surface soils
Future Commercial Workers:	Incidental ingestion of surface soils Dermal contact with surface soils

Exposures via Subsurface Soil

Inhalation of VOCs from subsurface soils are potentially of concern to populations that may be exposed to these soils through excavation or other exposure scenarios. Inhalation of VOCs from subsurface soil is a potential pathway for trespassers, residents, construction workers, and commercial workers. However, as discussed in Section 6.2.3, VOCs were not found to be of concern in subsurface soil at the site. Further, VOCs were not determined to be COPCs using surface or total soil data.

Exposures via Total Soil

The following exposure pathways are considered to be complete for total soil at Site PI-1 (Total soil includes surface and subsurface soil):

Future Onsite Resident (adult, child):	Incidental ingestion of total soils Dermal contact with total soils
Future Construction Worker:	Incidental ingestion of total soils Dermal contact with total soils

Exposures via Groundwater

The following exposure pathways are considered to be complete for onsite groundwater.

Future Onsite Resident (adult, child):	Ingestion of groundwater Dermal contact in shower (adult) and in bath (child)
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Exposures via Air

Exposures to airborne particles from soils were evaluated for the following scenarios/receptors:

Current Trespasser (adult, adolescent):	Inhalation of particles entrained from surface soil
Future Onsite Resident (adult, child):	Inhalation of particles entrained from total soil

Future Construction Worker:	Inhalation of particles entrained from total soil
Future Commercial Workers:	Inhalation of particles entrained from surface soil.

6.2.2 Risk-based Screening

Risk-based screening was conducted by comparing maximum detected chemical concentrations for each medium to risk-based screening concentrations. An analyte in a medium for which the maximum measured concentration exceeded the risk-based screening concentration was retained as a COPC.

This risk assessment is based on several sources of data as discussed in Chapter 4 and below. After submission of the Final Phase I RI (EA 2000a), validated soil data from WESTON (1998) was incorporated in the HHRA per recommendation of U.S. EPA (EPA 2000). The WESTON (1998) data included six surface soil samples in which inorganic compounds were detected above screening limits. The organic compounds that were detected were at concentrations below screening criteria. These six samples were added to the RI database and the HHRA was revised starting with the screening assessment. In this process, the maximum detected concentration, the arithmetic mean and 95% upper confidence limit on the mean (95%UCLM) were revised based upon the additional sample results for all inorganics. As a result, the analysis of inorganic analytes in this assessment includes the WESTON data while the organic analyte analysis does not.

Risk-based screening concentrations that were used in the selection of COPCs are medium specific and are discussed below. For soils, the U.S. EPA Region III residential soil RBC (EPA 1999a) were used for residential screening (resident adult, resident child, adult trespasser, and adolescent trespasser); for industrial screening the industrial soil RBC was used (commercial worker and construction worker); and for groundwater, the lower of the federal MCL or the Region III tap water RBCs (EPA 1999a) was used. Per Region III guidance (EPA 1993a), one-tenth of the RBC for non-carcinogens was used to account for potential additivity of toxic effects.

An analyte was eliminated from the list of COPCs if it was an essential nutrient of low toxicity, and if its reported maximum concentration was unlikely to be associated with adverse health impacts. COPCs excluded from further consideration on this basis included calcium, magnesium, potassium, and sodium. While iron can also be considered an essential nutrient,

Surface soil COPCs (Table 6-2.3) based on residential soil RBC screening included aluminum, arsenic, chromium, iron, thallium, vanadium, and dioxin. The surface soil COPCs (Table 6-2.4) based on industrial soil RBCs were arsenic and dioxin.

Subsurface soil screening is presented for residential and industrial RBCs in Tables 6-2.5 and 6-2.6; subsurface soils were evaluated to assess the potential for exposure via inhalation to VOCs only. None of the VOCs exceeded RBCs for either scenario; therefore there are no volatile COPCs with respect to the inhalation route in subsurface soil.

Groundwater COPCs are presented in Table 6-2.7. As discussed in Section 4.4.3 and per Region III guidance (EPA 1992a), filtered data were used to represent inorganic groundwater concentrations due to the disparity between filtered and unfiltered data results. Groundwater COPCs were determined in a conservative manner via a screen against tap water RBCs. Manganese, heptachlor, aldrin, heptachlor epoxide, and alpha HCH were determined to be COPCs in groundwater. No VOCs were detected in groundwater; therefore, there are no risks associated with inhalation of groundwater.

6.3 EXPOSURE ASSESSMENT

An exposure assessment is conducted to estimate the magnitude of potential human exposures to COPCs in site media. In the exposure assessment, average and reasonable maximum estimates of potential exposure are developed in accordance with U.S. EPA guidance for both current and potential future land-use assumptions. Current reasonable maximum exposure estimates are used to determine whether a potential health hazard exists based on current conditions. Future potential exposure estimates are used to provide an understanding of potential future exposures and health hazards and include a qualitative estimate of the likelihood of such exposures occurring. Conducting an exposure assessment involves analyzing releases of chemicals of potential concern; identifying all potential pathways of exposure; estimating average and reasonable maximum potential exposure point concentrations for specific pathways, based both on environmental monitoring data and predictive chemical modeling results; and estimating potential chronic daily intakes for specific pathways. The results of this assessment are pathway-specific estimates of potential intakes for current and future exposures to individual chemicals of potential concern.

6.3.1 Quantification of Potential Exposures

The first step of the exposure assessment is to quantify potential exposure concentrations. This involves the evaluation of site data and the quantification of exposure concentrations for average and reasonable maximum exposure scenarios.

6.3.1.1 Data Quality Evaluation

Inclusion or exclusion of data on the basis of analytical qualifiers was performed in accordance with U.S. EPA guidance (EPA 1989b). Section 4.4 discusses data reduction for Site PI-1 in detail; however, highlights relating to human health risk assessment are presented here:

- Analytical results bearing the U qualifier (indicating that the analyte was not detected at the given sample quantitation level [SQL]) were retained in the data set and considered non-detects. Where warranted for statistical purposes, each COPC was assigned a numerical value of one-half its SQL;
- Analytical results bearing the J qualifier (indicating that the reported value was estimated because the analyte was detected at a concentration below the SQL or for other reasons) were retained at the measured concentration.

If duplicate samples were taken or duplicate analyses were conducted on a single sample, the following guidelines were employed to select the appropriate sample measurement:

- If both samples/analyses showed that the analyte was present, the average of the analyses was used as the concentration;
- If only one sample/analysis indicated that the analyte was present, it was retained for analysis; and combined with 1/2 the detection limit to calculate the average; and
- If both samples/analyses were non detect, the average of the 1/2 of the detection limits were retained for analysis, if appropriate.

Common laboratory contaminants, including acetone, 2-butanone, methylene chloride, chloroform, toluene, phthalate esters, and uncommon laboratory contaminants were considered to be COPCs unless it was evident that their presence was not related to site-specific activities

but were due to laboratory contamination. A detailed discussion regarding the treatment of blanks and associated B-qualified data specific to this site is provided in Section 4.4.

6.3.1.2 Analysis of COPC Data

To assess human health risks, the statistical analysis of the COPC concentrations in each medium were performed. The methods used to analyze the data for each of these media are described below.

Surface soil, total soil, and groundwater samples are potential site media of concern. There were no volatile COPCs identified in subsurface soil; therefore, inhalation of VOCs from subsurface soil is not a concern. For total soil, surface soil, and groundwater, reported concentrations were used to calculate the 95th percentile upper confidence limit on the mean (95UCLM) for COPCs in each medium (EPA 1992b). Exposure point concentrations (EPCs) in site media were estimated as the 95UCLM values for purposes of estimating reasonable maximum exposures. In cases where the 95UCLM values exceeded the maximum detected concentration, the maximum detected concentration was used. The first step in estimation of EPC is to determine whether medium-specific environmental data for a COPC is normally or log-normally distributed. This was accomplished with the Shapiro-Wilks W-test for distribution (Gilbert 1987). If the data fit neither the normal or log-normal distributions, based on comparison of the calculated W-statistic with critical W-statistic values, the distribution with the highest W-statistic was used. W-statistic results are presented in Table H-46.

For a log-normally distributed COPC the following steps were carried out to calculate 95UCLM. Because transformation is a necessary step in calculating the upper confidence limit of the mean (UCLM) for a log-normal distribution, the data were transformed by using the natural logarithm function (i.e., calculate $\ln(x)$, where x is the value from the data set). After transforming the data, 95UCLM for the data set was calculated by calculating the arithmetic mean of the transformed data; calculating standard deviation of the transformed data; determining H-statistic (Gilbert 1987); and calculating 95UCLM using the equation given below:

$$95\ UCLM = e^{(\bar{x} + 0.5\ s^2 + s\ H / \sqrt{n-1})} \quad (\text{Equation 6-1})$$

Where:

95UCLM	=	95th percentile upper confidence limit on the mean
e	=	Constant (base of the natural logarithm; equal to 2.718)

\bar{x}	=	Mean of the transformed data
s	=	Standard deviation of the transformed data
H	=	H-Statistic
n	=	Number of samples in the data set

If the statistical test supported the assumption that the data set for a COPC was normally distributed the following steps were undertaken to calculate 95UCLM (EPA 1992b):

(1) calculate the arithmetic mean of the untransformed data; (2) calculate standard deviation of the untransformed data; (3) determine the one-tailed t-statistic (Gilbert 1987); and (4) calculate 95UCLM using the equation given below:

$$95\text{ UCLM} = (\bar{x} + t s / \sqrt{n}) \quad (\text{Equation 6-2})$$

Where:

95UCLM	=	95th percentile upper confidence limit on the mean
\bar{x}	=	Mean of the untransformed data
s	=	Standard deviation of the untransformed data
t	=	Student-t statistic
n	=	Number of samples in the data set.

6.3.1.3 Exposure Point Concentrations

Tables 6-3.1 through 6-3.5 present summary statistics (e.g., frequency of detection, range of detection, mean, and the 95UCLM) for each COPC in total soil (residential and industrial); in surface soil (residential and industrial); and in groundwater respectively at the site. The reasonable maximum exposure (RME) EPC value was utilized as the chemical-specific, medium-specific EPC in the exposure assessment for the RME exposure assumptions. However, if the 95UCLM was greater than the maximum detected concentration, the maximum detected concentration value was used as the EPC and is listed in the table instead of the 95 UCLM value, as per U.S. EPA guidance (EPA 1989b). For average exposure (AE) assumptions, the arithmetic mean concentration was used. COPCs for the adult and adolescent trespasser (Table 6-3.3) were determined by screening surface soil against residential RBCs.

6.3.2 Exposure Equations

The next step in this exposure assessment was to estimate COPC intakes for each of the pathways considered in the assessment. In this exposure assessment, we have provided two different measures of intake, depending on the nature of the effect being evaluated. When evaluating longer-term (i.e., subchronic and chronic) exposures to chemicals that produce adverse non-carcinogenic effects, intakes are averaged over the period of exposure (i.e., the averaging time [AT]) (EPA 1989b). This measure of intake is referred to as the average daily intake (ADI) and is a less than lifetime exposure. For chemicals that produce carcinogenic effects, intakes are averaged over an entire lifetime and are referred to as the lifetime average daily intake (LADI) (EPA 1989b).

The generic equation to calculate intakes is given below:

$$(L)ADI = \frac{C \times IF \times EF \times ED \times RAF}{BW \times AT} \times CF \quad (\text{Equation 6-3})$$

Where:

<i>(L)ADI</i>	=	(Lifetime) Average daily dose (mg/kg-day)
<i>C</i>	=	Concentration in a specific medium (mg/L or mg/kg)
<i>IF</i>	=	Intake factor ¹ (mg/day)
<i>EF</i>	=	Exposure frequency (days/year)
<i>ED</i>	=	Exposure duration (years)
<i>RAF</i>	=	Relative absorption factor (unitless)
<i>BW</i>	=	Body weight (kg)
<i>AT</i>	=	Averaging time (days)
<i>CF</i>	=	Conversion Factor (10 ⁻⁶ kg/mg)

The exposure pathways which are considered to be complete for Site PI-1 are presented in Figure 6-1 and Table 6-1. The equations used to estimate exposures for each of the exposure pathways assessed for Site PI-1 are presented in Appendix H.

¹ The intake factor is the product of all intake variables that, when multiplied by the concentration of the chemical of potential concern in a specific medium, results in an estimate of the chemical intake in mg/kg-day for that population and exposure pathway. Intake factors may include ingestion rate, inhalation rate, body surface area exposed to soil or water, dermal permeability constants, and soil adherence factors.

6.3.3 Selection of Exposure Factor Values

All exposure factor values used in estimating intakes are described and referenced in Table 6-4.1 to 6-4.8. The following guidance documents were used in defining exposure factor values for estimating intakes for exposure pathways evaluated at Site PI-1:

- Risk Assessment Guidance for Superfund (RAGS) Volume I Human Health Evaluation Manual Part A, U.S. EPA December 1989 (EPA 1989b).
- Office of Solid Waste and Emergency Response (OSWER) Directive 9285.6-03; Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors", U.S. EPA, 1991 (EPA 1991).
- Exposure Factors Handbook, Volume I, General Factors, U.S. EPA, August 1997 (EPA 1997a).
- Dermal Exposure Assessment: Principles and Applications, U.S. EPA 1992 (EPA 1992c).
- Risk Assessment for Superfund, Volume 1: Human Health Evaluation Manual, Part E, Supplemental Guidance for Dermal Risk Assessment, Interim Guidance, U.S. EPA, 2000 (EPA 2000).
- Technical Guidance Manual, Risk Assessment, U.S. EPA Region III 1993, 1995 (EPA 1993a and 1995b).

For all exposure pathways which have exposure factor values specified in RAGS Part A and in OSWER Directive 9285.6-03, those values were used in this risk assessment. For exposure values not specified in the RAGS Part A or the OSWER Directive, the Region III Guidance documents, and the Exposure Factors Handbook were used to determine appropriate exposure values, where available. U.S. EPA Dermal Guidance documents from 1992 and 2000 were also utilized. All exposure factor values utilized are presented in the following sections.

6.3.3.1 Future Residents

Future residential users may potentially be exposed to COPCs via total soil and groundwater. COPCs in total soil for future residents were determined based upon screening against residential RBCs.

Both adults and children were assessed for the residential scenario. Children are considered sensitive receptors relative to adult receptors based on several factors. Children have a lower body weight than adults and are therefore exposed to a higher concentration per kilogram than adults. In addition, childhood development and growth can increase metabolism of certain chemicals and may create a greater effect in children than in adults. Therefore, the residential scenario is evaluated for an adult and a child to capture the potential sensitivity of the child receptor. To evaluate lifetime carcinogenic risks, the combination of the adult and child residential receptors results in an approximation of age-adjusted lifetime cancer risk.

Residential Adults - Exposure parameters for residential adult exposure are presented in Table 6-4.1 for total soil and in Table 6-4.7 for groundwater. Body weight for the adult resident was assumed to be 70 kg. Under RME conditions, future adult residents were assumed to have an exposure duration of 30 years for non-carcinogenic risk and 24 years for carcinogenic risks. An exposure frequency of 350 days/year was used. Surface area available for dermal exposure to total soil was assumed to be 5,700 cm² with an adherence factor of 0.07. An inhalation rate of 0.83 m³/hr for 24 hr/day was conservatively assumed. The residential adult RME groundwater ingestion rate was assumed to be 2 liters per day. Surface area exposed to groundwater while bathing/showering was assumed to be 18,000 cm² for 0.58 hour (35 minutes) exposure time.

Under AE conditions, future adult residents were assumed to have an exposure duration of 7 years with an exposure frequency of 150 days/year. Surface area available for dermal exposure to total soil was assumed to be 5,700 cm² with an adherence factor of 0.01. An inhalation rate of 0.83 m³/hr for 10 hr/day was assumed. The AE groundwater ingestion rate was assumed to be 1.4 liters per day. Surface area exposed to groundwater while bathing/showering was assumed to be 18,000 cm² for 0.25 hour (15 minutes) exposure time.

Residential Children - Exposure parameters for child resident exposure are presented in Table 6-4.2 for total soil and in Table 6-4.8 for groundwater. Body weight for the future child resident was assumed to be 15 kg. Under RME conditions, future child residents were assumed to have a 6-year exposure duration with an exposure frequency of 350 days/year. RME surface area

available for dermal exposure to total soil was assumed to be $2,800 \text{ cm}^2$ with an adherence factor of 0.2. An inhalation rate of $0.417 \text{ m}^3/\text{hr}$ for 24 hr/day was assumed. The residential child RME groundwater ingestion rate was assumed to be 1 liter per day. RME surface area exposed to groundwater while bathing was assumed to be $6,600 \text{ cm}^2$ for 1.0 hour exposure time.

Under AE conditions, future child residents were assumed to have a 2-year exposure duration with an exposure frequency of 150 days/year. Surface area available for dermal exposure to soil was assumed to be $2,800 \text{ cm}^2$ with an adherence factor of 0.06. An inhalation rate of $0.417 \text{ m}^3/\text{hr}$ for 10 hr/day was assumed. The AE groundwater ingestion rate was assumed to be 0.6 liter per day. AE surface area exposed to groundwater while bathing was assumed to be $6,600 \text{ cm}^2$ for 0.33 hour (20 minutes) exposure time.

6.3.3.2 Current Adult Trespasser

Exposure parameters for adult trespasser exposure are presented in Table 6-4.3 for surface soil. COPCs in surface soil for the adult trespasser were determined by conservatively screening against residential RBCs. Body weight for the adult trespasser was assumed to be 70 kg. Under RME conditions, adult trespassers were assumed to have an exposure duration of 30 years to correspond to the time period in which an adult is expected to remain within one residence. An exposure frequency of 39 days/year was assumed based on best professional judgement. A surface soil ingestion rate of 100 mg/d was assumed. Dermal exposure to soil was based on $5,700 \text{ cm}^2$ surface area and $0.07 \text{ mg}/\text{cm}^2$ adherence factor. The RME inhalation rate was assumed to be $0.83 \text{ m}^3/\text{hour}$ for particulate over an exposure time of 8 hours/day.

Under AE conditions, adult trespassers were assumed to have an exposure duration of 1 year with an exposure frequency of 8 days/year. A surface soil ingestion rate of 50 mg/d was assumed. AE dermal exposure to surface soil was based on $5,700 \text{ cm}^2$ surface area and $0.01 \text{ mg}/\text{cm}^2$ adherence factor. The RME inhalation rate was assumed to be $0.83 \text{ m}^3/\text{hour}$ for particulate over an exposure time of 4 hours/day.

6.3.3.3 Current Adolescent Trespasser

Exposure parameters for adolescent trespasser exposure are presented in Table 6-4.4 for surface soil. COPCs in surface soil for the adolescent trespasser were determined by conservatively screening against residential RBCs. The age range of the adolescent trespasser was assumed to be 12 to 18 years, and the body weight for the adolescent trespasser was assumed to be 36 kg.

Under RME conditions, adolescent trespassers were assumed to have an exposure duration of 6 years with an exposure frequency of 141 days/year. The exposure frequency is based on an average of the mean days of outdoor activity for a young child (130days/year) and an older child (152 days/year). (EPA 1997) A surface soil ingestion rate of 200 mg/d was assumed. RME dermal exposure to soil was based on 2,900 cm² surface area and 0.08 mg/cm² adherence factor. The RME inhalation rate was assumed to be 0.83 m³/hour for particulate over an exposure time of 24 hours/day.

Under AE conditions, adolescent trespassers were assumed to have an exposure duration of 1 year with an exposure frequency of 10 days/year. A surface soil ingestion rate of 100 mg/d was assumed. RME dermal exposure to soil was based on 2,085 cm² surface area and 0.08 mg/cm² adherence factor. The RME inhalation rate was assumed to be 0.83 m³/hour for particulate over an exposure time of 4 hours/day.

6.3.3.4 Construction Worker

Exposure parameters for construction worker exposure are presented in Table 6-4.5 for total soil. COPCs in total soil for the construction worker were determined by screening against industrial RBCs. Body weight for the construction worker was assumed to be 70 kg. Under RME conditions, future construction workers were assumed to have an exposure duration of 1 year with an exposure frequency of 150 days/year. Skin surface area available for contact with total soil during construction activities was assumed to be 3,300 cm² with an adherence factor of 0.03 mg/cm². Incidental ingestion of soil was assumed to be 480 mg/day. The RME inhalation rate was assumed to be 0.83 m³/hour for total soil particulate over an exposure time of 8 hours per day.

Under AE conditions, future construction workers were assumed to have an exposure duration of 1 year with an exposure frequency of 10 days/year. Skin surface area available for contact with total soil during construction activities was assumed to be 3,300 cm² with an adherence factor of 0.1 mg/cm². Incidental ingestion of soil was assumed to be 100 mg/day. The AE inhalation rate was assumed to be 0.83 m³/hour for particulate over an exposure time of 8 hours/day.

6.3.3.5 Commercial Worker

Exposure parameters for commercial worker exposure are presented in Table 6-4.6 for surface soil. COPCs in surface soil for the commercial worker were determined by screening against industrial RBCs. Body weight for the commercial worker was assumed to be 70 kg. Under

RME conditions, commercial workers were assumed to have an exposure duration of 25 years with an exposure frequency of 250 days/year. A surface soil ingestion rate of 50 mg/d was assumed. Dermal exposure to soil was based on 3,300 cm² surface area and 0.2 mg/cm² adherence factor. The RME inhalation rate was assumed to be 0.83 m³/hour for particulate over an exposure time of 8 hours/day.

Under AE conditions, commercial workers were assumed to have an exposure duration of 8 years with an exposure frequency of 165 days/year. A surface soil ingestion rate of 25 mg/d was assumed. AE dermal exposure to surface soil was based on exposed 3,300 cm² surface area and 0.02 mg/cm² adherence factor. The AE inhalation rate was assumed to be 0.83 m³/hour for particulate over an exposure time of 4 hours/day.

6.4 TOXICITY ASSESSMENT

The toxicity assessment considers the types of potential adverse health effects associated with exposures to COPCs; the relationship between magnitude of exposure and potential adverse effects; and related uncertainties, such as the weight of evidence of a particular COPC's carcinogenicity in humans. The toxicity assessment for COPCs relies on existing toxicity information developed on specific organic compounds and inorganic constituents. U.S. EPA Guidance (EPA 1989b) specifies that the assessment is accomplished in two steps: hazard identification and dose-response assessment. Hazard identification is the process of determining whether studies claim that exposure to a COPC may cause the incidence of an adverse effect. U.S. EPA specifies the dose-response assessment which involves: (1) U.S. EPA's quantitative evaluation of the existing toxicity information, and (2) U.S. EPA's characterization of the relationship between the dose of the COPC administered or received, and the incidence of potentially adverse health effects in the exposed population. From this quantitative dose-response relationship, specific toxicity values are derived by U.S. EPA which can be used to estimate the incidence of potentially adverse effects occurring in humans at different exposure levels (EPA 1989b). These U.S. EPA-derived toxicity values are called RfDs for non-carcinogens and SFs for potential carcinogens. The toxicity values used for COPCs at Site PI-1 are presented in Tables 6-5.1 and 6-5.2 for non-carcinogens and in Table 6-6.1 and 6-6.2 for carcinogens. Toxicological profiles for all COPCs are presented in Appendix G.

6.4.1 Toxicity Assessment for Non-Carcinogens

For all COPCs, toxicity values for non-carcinogens were taken, when available, from the Integrated Risk Information Systems (IRIS) database (EPA 1999b). IRIS chronic toxic potency concentrations are developed by U.S. EPA and undergo an extensive process of scientific peer review. Therefore, IRIS values are judged to be adequately verified.

If toxic potency concentrations for COPCs were not available from IRIS (EPA 1999b), EPA (1997b) health effects assessment summary tables (HEAST) was used as a secondary data source. As HEAST toxicity values are not scientifically peer-reviewed for quality or scientific acceptability, they may not be derived in strict accordance with U.S. EPA-approved methodologies.

If IRIS or HEAST toxic potency concentrations were not available for one route of exposure but existed for another route, the existing value was examined for technical applicability to the alternate route and subsequently utilized, if appropriate.

The methodology used by U.S. EPA for deriving toxic potency concentrations for non-carcinogens, as well as site-specific considerations for modifying or using these concentrations are discussed in detail in Barnes and Dourson (1988) and U.S. EPA guidance (EPA 1989b). Non-carcinogens are typically judged to have a threshold daily dose below which deleterious or harmful effects are unlikely to occur. This concentration is called the no-observed-adverse-effect-level (NOAEL) and may be derived from either animal laboratory experiments or human epidemiology investigations (usually workplace studies). In developing a toxicity value or human NOAEL for non-carcinogens (i.e., an RfD), the regulatory approach is first to (1) identify the critical toxic effect associated with chemical exposure (i.e., the most sensitive adverse effect); (2) identify the threshold dose in either an animal or human study; and (3) modify this dose to account for interspecies variability (where appropriate), differences in individual sensitivity (within-species variability), and other uncertainty and modifying factors. Uncertainty factors are intended to account for specific types of uncertainty inherent in extrapolation from the available data. Modifying factors account for the concentration of confidence in the scientific studies from which toxicity values are derived, according to such parameters as study quality and study reproducibility. The use of these factors is a conservative approach to protection of human health and is likely to overestimate the toxic potency associated with chemical exposure. The resulting RfD is expressed in units of milligrams of chemical per kilogram of body weight per day (mg/kg-bw/day).

Toxicity values used for exposures that involve dermal contact with chemicals typically require adjustment of the oral toxicity values (oral RfDs) to allow for the difference between the daily intake dose through dermal contact and ingestion. Most toxicity values are based on the actual administered dose, and must be corrected for the percent of chemical-specific absorption that occurs across the gastrointestinal tract prior to their use in dermal contact risk assessment (U.S. EPA 1989b, 1992c and 1998a). Region III recommended oral absorption efficiency factors were utilized in converting oral toxicity values to dermal toxicity values (EPA 1995b). These factors are shown in Table 6-5.3.

6.4.2 Toxicity Assessment for Carcinogenicity

Unlike non-carcinogens, carcinogens are generally assumed to have no threshold, that is, there is presumed to be no level of exposure below which carcinogenic effects will not manifest themselves. This “non-threshold” concept supports the idea that there are small, finite probabilities of inducing a carcinogenic response associated with every level of exposure to a potential carcinogen. U.S. EPA uses a two-part evaluation for carcinogenic effects, which includes the assignment of a weight-of-evidence classification to a chemical based on a thorough scientific examination of the body of available data, and the quantification of a cancer toxic potency concentration, i.e., the slope factor, which reflects the dose-response data for the carcinogenic endpoint(s) (EPA 1989b).

The weight-of-evidence classification system assigns a letter or alphanumeric (A through E) to each potential carcinogen that reflects an assessment of its potential to be a human carcinogen.² Only compounds that have a weight-of-evidence classification of C or above are considered to have carcinogenic potential in this risk assessment.

Although currently a controversial approach, chemicals that are classified as human or rodent carcinogens are typically assumed to have no threshold, in that there is presumed to be no concentration of exposure below which carcinogenic effects will not be manifested. The U.S. EPA slope factor is the upper 95th percentile confidence limit of the probability of response per unit daily intake of a chemical over a lifetime. Typically, the slope factor is used to estimate the

²A = a known human carcinogen; B1 = a probable human carcinogen, based on sufficient animal data and limited human data; B2 = a probable human carcinogen based on sufficient animal data and inadequate or no human data; C = a possible human carcinogen; D = not classifiable as to human carcinogenicity; and E = evidence of non-carcinogenicity for humans.

upper-bound lifetime probability of a person developing cancer from exposure to a given concentration of a carcinogen. Slope factors are generally based on experimental animal data, unless suitable epidemiological studies are available. Due to the difficulty in detecting and measuring carcinogenic endpoints at low exposure concentrations, slope factors are typically developed by using a model to fit the available high-dose, experimental animal data, and then extrapolating downward to the low-dose range to which humans are typically exposed. U.S. EPA usually employs the linear multistage model, to derive a slope factor. The model is conservative, and provides an upper bound estimate of excess lifetime cancer risk. Thus, the actual risk may be lower and could be zero (EPA 1989b). These methods and approaches are discussed in greater detail in the U.S. EPA *Risk Assessment Guidance for Superfund* (EPA 1989b).

Carcinogenic slope factors used for exposures that involve dermal contact typically require adjustment of the oral slope factor to allow for the difference between the dermal dose and the ingested dose. Most toxicity values are based upon the actual administered dose, and must be corrected for the percent of chemical-specific absorption that occurs across the gastrointestinal tract prior to their use in dermal contact risk assessments (EPA 1989b). For inhalation exposures, inhalation slope factors are developed if sufficient data are available.

6.4.3 Toxicity of Constituents of Potential Concern (COPCs)

Toxicity data available in the scientific literature were used to prepare a toxicity profile for each COPC. Each profile describes the potential for carcinogenicity and other health effects of each COPC, summarizes available data, presents a weight-of-evidence approach for identifying the hazards associated with chemical exposure to the COPC, and provides a scientific profile for selecting the most appropriate toxicity values (i.e., quantitative estimates of the strength of the dose-response) used later in the risk assessment.

A review of relevant toxicity data for each COPC is presented in Appendix G, along with a description of critical studies (i.e., studies from which the quantitative toxic potency values were derived). Toxicity values were obtained from the U.S. EPA IRIS (EPA 1999b), a peer-reviewed toxicity database. If toxicity values were not available from IRIS, values from the U.S. EPA HEAST (EPA 1997b) or from the National Center for Environmental Assessment (NCEA) were used.

6.4.3.1 Summary of Toxicity Values for Non-Carcinogenic Effects

U.S. EPA-derived toxicity values for evaluating potential chronic non-carcinogenic effects for COPCs are summarized in Tables 6-5.1 and 6-5.2. Toxicity information presented in these tables includes the following U.S. EPA provided/derived information: chronic or subchronic RfD values for exposures via the oral and inhalation pathway; reported health effects, uncertainty and modifying factors specific to the U.S. EPA-derived RfD; and the scientific source of the information.

6.4.3.2 Summary of Toxicity Values for Potential Carcinogenic Effects

U.S. EPA-derived toxicity values for evaluating potential carcinogenic effects for COPCs are summarized in Tables 6-6.1 and 6-6.2. Toxicity information presented in these tables includes the following U.S. EPA provided/derived information: a chemical specific slope factor (cancer potency factor) for exposures via the oral and inhalation pathway; U.S. EPA's weight-of-evidence cancer classification; and the scientific source of the information.

6.4.3.3 Toxicity of Specific Chemicals of Concern at PI-1

Several chemicals are classified by U.S. EPA as B2, probable human carcinogens, and therefore, were conservatively evaluated as carcinogens in this HHRA. It should be noted that the carcinogenic toxicity values for these chemicals are not based on human data but are, instead, based on animal data. For example, the US EPA IRIS database lists the human carcinogenic data for aldrin and alpha-HCH as inadequate (EPA 1999b). The uncertainty associated with the evaluation of these chemicals is discussed in Section 6.8.3.3.

The toxicity values for dioxin are derived from studies based on one chemical, 2,3,7,8-tetrachlorodibenzodioxin. All other dioxin congeners are evaluated based on their relative toxicity to 2,3,7,8-tetrachlorodibenzodioxin to derive toxicity equivalence factors (TEFs) by the World Health Organization (WHO 1997). The TEFs are applied by the laboratory to concentrations of all detected congeners to estimate a dioxin concentration for each sample. One-half the detection limit is used for non-detected congeners. These TEF weighted concentrations are summed to estimate a total TEQ, or total dioxin concentration. The laboratory calculated total dioxin concentration was used in the HHRA and is evaluated as 2,3,7,8-tetrachlorodibenzodioxin. Toxicity values for 2,3,7,8-tetrachlorodibenzodioxin are taken from HEAST.

6.5 RISK CHARACTERIZATION

Risk characterization is the final step of the HHRA process. In this step, the toxicity values were combined with the estimated chemical intakes for the receptor populations to quantitatively estimate both carcinogenic risks and risks for non-carcinogens. Risks were estimated for the following receptor populations:

- Residents (Adult, Child)
- Trespassers (Adult, Adolescent)
- Commercial Workers
- Construction Workers

The methodologies used to estimate cancer risks and chronic and subchronic risks for non-carcinogens are described further in the sections below.

6.5.1 Hazard Index for Noncarcinogenic Effects

The potential human health risks associated with exposures to non-carcinogenic COPCs at EBS-79 were estimated by comparing the ADI with the RfD, as per U.S. EPA Guidance (EPA 1989b). An HQ was derived for each COPC, as shown in the equation below:

$$HQ = \frac{ADI}{RfD} \quad \text{(Equation 6-4)}$$

Where:

<i>HQ</i>	=	Hazard Quotient; ratio of average daily intake level to acceptable daily intake level (unitless)
<i>ADI</i>	=	Estimated average daily dose (mg/kg-day)
<i>RfD</i>	=	Reference dose (mg/kg-day)

If the average daily dose exceeds the RfD, the HQ will exceed one and there may be concern that potential adverse systemic health effects will be observed in the exposed populations. If the ADI does not exceed the RfD, the HQ will not exceed one and there will be no concern that potential adverse systemic health effects will be observed in the exposed populations. However, if the sum of several HQs exceeds one, and the COPCs affect the same target organ, there may be

concern that potential adverse systemic health effects will be observed in the exposed populations. In general, the greater the value of the HQ above one, the greater the level of concern. However, the HQ does not represent a statistical probability that an adverse health effect will occur.

For consideration of exposures to more than one chemical causing systemic toxicity via several different pathways, the individual HQs are summed to provide an overall hazard index (HI). If the HI is less than 1.0, then no adverse health effects are likely to be associated with exposures at the site. However, if the total HI is greater than 1.0, separate endpoint-specific HIs may be calculated based on toxic endpoint of concern or target organ (e.g., HQs for neurotoxins are summed separately from HQs for renal toxins). Only if an endpoint-specific HI is greater than one is there reason for concern about potential health effects for that endpoint.

6.5.2 Cancer Risks

Carcinogenic risk was estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen at the site. The numerical estimate of excess lifetime cancer risk was calculated by multiplying the LADI by the risk per unit dose (the slope factor), as shown in the following equation:

$$Risk = LADI \times SF \quad \text{(Equation 6-5)}$$

Where:

<i>Risk</i>	=	The unitless probability of an exposed individual developing cancer
<i>LADI</i>	=	Lifetime average daily dose (mg/kg-day)
<i>SF</i>	=	Cancer slope factor (mg/kg-day) ⁻¹

Because the slope factor is the statistical 95th percent upper-bound confidence limit on the dose-response slope, this method provides a conservative, upper-bound estimate of risk.

Cancer risks were estimated for current and future occasional users, for potential future residents, and for recreational hunters. It should be noted that the interpretation of the significance of the cancer risk estimate is based on the appropriate public policy. The U.S. EPA in the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (40 Code of Federal Regulations (CFR) Part 300) (EPA 1990) states that:



“...For known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} .”

The Commonwealth of Virginia Department of Environmental Quality (DEQ) goal is that the acceptable cancer risk range is below 10^{-6} for any one COPC (including all exposure pathways) or below 10^{-4} for all COPCs and pathways combined. Per conversation with DEQ (VA DEQ 2001), the results of the risk assessment will address this policy.

For the purposes of this risk assessment, the U.S. EPA definition of acceptable carcinogenic risk, the 10^{-4} to 10^{-6} range, will be applied. However, the DEQ policy will be discussed where more stringent than the U.S. EPA definition.

6.6 RISK RESULTS

Chemical-specific, pathway-specific risk estimates are presented by receptor group in Tables 6-7.1 through 6-12.4 and calculations are presented in Appendix H. Estimates of cumulative risks across pathways for non-carcinogenic and carcinogenic effects for each receptor population at Site PI-1 are presented in Tables 6-13 and 6-14. For the residential scenario, estimated risks were summed for the adult and child receptors to assess lifetime cancer risks. Combination of the adult and child residential receptors results in an approximation of age-adjusted lifetime cancer risk.

For non-cancer risks, the cumulative Hazard Index (HI) only exceeded the acceptable threshold of 1.0 for potential future resident child (Table 6-13). The ingestion of groundwater and the ingestion of total soil pathways drove these risks. Cumulative HIs of 3.6 under RME and 1.2 under AE assumptions were estimated for resident children.

Cumulative cancer risk estimates, which are presented in Table 6-14, did not exceed the U.S. EPA's “acceptable risk range” of 10^{-6} to 10^{-4} for any receptor. In addition, although some individual chemicals did exceed the DEQ goal of 10^{-6} for individual chemicals, none of the receptors were found to have an unacceptable cumulative cancer risk above 10^{-4} . The estimated risks were within this range for potential future residents (combined adult and child), adult trespassers, adolescent trespassers, and commercial workers. The cumulative cancer risks for construction workers were below this range.

The results for each receptor population are presented in detail in the following sections.

6.6.1 Future Onsite Resident Adult

The cancer and non-cancer risk assessment results for potential future onsite resident adults for AE and RME assumptions are presented in Tables 6-7.1 through 6-7.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H.

Total RME and AE non-cancer risks across all pathways and media for adult residents were 0.76 and 0.44, respectively, below the 1.0 threshold. Thus, no non-carcinogenic concerns for adult residents are present at the site.

Excess lifetime cancer risks to future onsite resident adults are evaluated as part of the total lifetime residential risk results; however, the individual adult cancer risks are shown in Tables 6-7.3 and 6-7.4 for RME and AE, respectively. The future lifetime cancer risks for residents are discussed below and are shown in Tables 6-8.5 and 6-8.6.

6.6.2 Future Onsite Resident Child

The cancer and non-cancer risk assessment results for potential future residential children for AE and RME scenarios are presented in Tables 6-8.1 through 6-8.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H. Children are considered sensitive receptors relative to adult receptors based on several factors. Children have a lower body weight than adults and are therefore exposed to a higher concentration per kilogram than adults. In addition, childhood development and growth can increase metabolism of certain chemicals and may create a greater effect in children than in adults. The RME and AE HIs across all pathways and media for the resident child were calculated as 3.6 and 1.2, respectively.

Potential future residential exposures to children resulting from total soil result in a cumulative HI of 2.1 for RME and 0.32 for AE assumption. Under RME, there were no individual analytes in total soil with HQ exceeding 1.0. Several analytes had HQ exceeding 0.1: aluminum, arsenic, chromium, iron, manganese, thallium, and vanadium. A breakdown by target organ is presented in Table 6-8.3. No target organ had an HI greater than 1.0. Therefore, there are no concerns for adverse non-cancer health effects for potential future child residents due to contact with site soil.

Potential future residential exposures to children resulting from ingestion of and dermal contact with groundwater result in a cumulative HI of 1.0 under RME and 0.84 under AE assumptions. Under RME the only individual COPCs that exceeded the 1.0 threshold was manganese. The only COPC which exceeded $HQ=0.1$ was aldrin. A breakdown by target organ is provided in Table 6-8.7. Neither of these COPCs has the same target organ. Therefore, aside from manganese (target organ is the central nervous system), the target organ specific HQs are less than 1.0, and there are no concerns for adverse non-cancer health effects for children from residential groundwater use.

Excess lifetime cancer risks to future onsite resident children are evaluated as part of the total lifetime residential risk results; however, the individual child cancer risks are shown in Tables 6-8.3 and 6-8.4 for RME and AE, respectively. The future lifetime cancer risks for residents are discussed below and are shown in Tables 6-8.5 and 6-8.6.

6.6.3 Future Onsite Resident (Adult and Child)

Excess lifetime cancer risks to future residents (combined adult and child) across all media and pathways were within U.S. EPA's acceptable risk range of 10^{-6} to 10^{-4} for cumulative risks and are presented in Tables 6-8.5 and 6-8.6. The RME excess cancer risk for the resident across all pathways and media were calculated as 7.7×10^{-5} and as 5.0×10^{-6} for the AE scenario.

The total soil-based total excess lifetime cancer risk was 3.2×10^{-5} under RME and 7.9×10^{-7} under AE conditions. The only RME COPCs with risks exceeding 10^{-6} were arsenic and dioxin.

The groundwater-based total excess lifetime cancer risk was 4.5×10^{-5} under RME and 4.1×10^{-6} under AE conditions. The RME COPCs with risks exceeding 10^{-6} were aldrin, alpha HCH, heptachlor, and heptachlor epoxide.

Arsenic and dioxin in soil and aldrin, alpha HCH, heptachlor and heptachlor epoxide in groundwater had calculated RME risks exceeding 10^{-6} which is the DEQ individual chemical threshold goal for acceptable risks. However, cumulative risks for this pathway were below the DEQ threshold of 10^{-4} .

6.6.4 Current Adult Trespasser

The cancer and non-cancer risk assessment results for adult trespassers for AE and RME scenarios are presented in Tables 6-9.1 through 6-9.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H.

Potential exposures to adult trespassers resulting from contact with surface soil result in a cumulative HI of 0.003 for RME and 0.0002 for AE assumption. Under RME, there were no individual analytes with HQ exceeding 0.1. Therefore, there are no concerns for adverse non-cancer health effects for potential adult trespassers at the site.

Excess lifetime cancer risks to adult trespassers resulting from surface soil contact were below U.S. EPA's acceptable risk range of 10^{-6} to 10^{-4} for cumulative risks under the specified RME conditions. The total excess lifetime cancer risk was 4.5×10^{-7} under RME and 1.9×10^{-9} under AE conditions. Therefore, there are no concerns for cancer risks resulting from site use by adult trespassers under current site conditions.

Arsenic and dioxin in soil and aldrin, alpha HCH, heptachlor and heptachlor epoxide in groundwater had calculated RME risks exceeding 10^{-6} which is the DEQ individual chemical threshold for acceptable risks. However, cumulative risks for this pathway were below the DEQ threshold of 10^{-4} .

6.6.5 Current Adolescent Trespasser

The cancer and non-cancer risk assessment results for adolescent trespassers for AE and RME scenarios are presented in Tables 6-10.1 through 6-10.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H.

Potential exposures to adolescent trespassers resulting from contact with surface soil result in a cumulative HI of 0.04 for RME and 0.0009 for AE assumption. Under RME, there were no individual analytes with HQ exceeding 0.1. Therefore, there are no concerns for adverse non-cancer health effects for potential adolescent trespassers at the site.

Excess lifetime cancer risks to adolescent trespassers resulting from surface soil contact were within U.S. EPA's acceptable risk range of 10^{-6} to 10^{-4} for cumulative risks under the specified RME conditions. The total excess lifetime cancer risk was 3.5×10^{-6} under RME and 8.8×10^{-9} under AE conditions. Arsenic and dioxin risks exceeded 10^{-6} (the DEQ goal) for the incidental

ingestion of surface soil pathway only; however, the cumulative carcinogenic risks were below the DEQ goal.

6.6.6 Future Construction Worker

The cancer and non-cancer risk assessment results for future construction workers for AE and RME scenarios are presented in Tables 6-11.1 through 6-11.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H.

The analysis presented in this report shows that construction worker exposures to COPCs in total soil result in a cumulative hazard index less than 1.0. Using RME, total HI was 0.21 for future construction workers under RME assumptions and 0.002 under AE assumptions, indicating there are no concerns for non-cancer health effects for future construction workers at the site.

Cumulative excess lifetime cancer risks to future construction workers fell below U.S. EPA's risk target of 10^{-6} , thus they were also under the DEQ goal for cumulative chemicals of 10^{-4} and for individual chemicals of 10^{-6} . Under RME assumptions, cancer risk estimates were 7.4×10^{-7} and 4.3×10^{-9} under AE assumptions. Therefore, there are no concerns for adverse health effects resulting from exposures to carcinogenic COPCs for future construction workers at the site under current or future use conditions.

6.6.7 Future Commercial Worker

The cancer and non-cancer risk assessment results for commercial workers for AE and RME scenarios are presented in Tables 6-12.1 through 6-12.4 and are summarized in Tables 6-13 and 6-14. The exposure scenario assumptions and calculations are presented in Appendix H.

The analysis presented in this report shows that commercial worker exposures to COPCs in surface soil result in a cumulative hazard index that is less than 1.0. Using RME, total HI was 0.01 for future construction workers under RME assumptions and 0.002 under AE assumptions, indicating there are no concerns for non-cancer health effects for future construction workers at the site.

Excess lifetime cancer risks to commercial workers resulting from soil exposure were within U.S. EPA's acceptable risk range of 10^{-6} to 10^{-4} for cumulative risks under the specified RME conditions. The total excess lifetime cancer risk was 4.6×10^{-6} under RME and 1.5×10^{-7} under AE conditions. The only individual COPC with risks exceeding 10^{-6} (the DEQ goal for

individual chemicals) was dioxin. However, cumulative chemical carcinogenic risks were below the DEQ goal of 10^{-4} .

6.7 BACKGROUND SOIL DATA COMPARISON

A background soil survey was conducted at Fort Pickett in November 1999 (WESTON 1999a). This survey established background concentration ranges for Fort Pickett for inorganics, PAHs, pesticides, and PCBs. For the purposes of this human health risk assessment, inorganic COPCs in soil were compared to the results of the background study.

The inorganic COPCs in soil at PI-1 were aluminum, arsenic, chromium, manganese, thallium, and vanadium. With the exception of thallium, which was not detected in the background study, all of these constituents were detected within the range of detected background concentrations in site soils. The maximum detected site-related concentration in soils for each of these COPCs was below the maximum detected background soil concentration. Table 6-15 provides a comparison of inorganic background detections to site-related detections for soil COPCs.

Manganese and arsenic concentrations in soil were determined to be risk drivers based on the calculations of risk for potential future resident children. However, the maximum detected concentrations of both of these constituents is consistent with site background. Manganese background detections ranged from 2.5 to 1,270 mg/kg. The maximum detected site-related manganese concentration was 399 mg/kg. The manganese 95UCLM based on background data was 735 mg/kg compared to the site-related total soil 95UCLM of 184 mg/kg. The manganese mean based on background data was 217 mg/kg compared to the site-related total soil mean of 120 mg/kg. Therefore, it is reasonable to assume that risk attributed to manganese is due to background and is not related to previous activities at Site PI-1.

Detected arsenic background concentrations ranged from 0.91 to 6.0 mg/kg. The maximum detected site-related arsenic concentration was 3.1 mg/kg, which was less than the maximum detected background concentration. The arsenic 95UCLM based on background data was 3.69 mg/kg compared to the site-related total soil 95UCLM of 1.85 mg/kg. The arsenic mean based on background data was 2.71 mg/kg compared to the site-related total soil mean of 1.4 mg/kg. Therefore, it is reasonable to assume that risks attributed to arsenic is due to background and is not related to previous activities at Site PI-1.

Background data for groundwater were not available for the site. However, EA evaluated offsite groundwater quality data, including manganese levels, from other wells located at Fort Pickett. Regarding the pesticide and manganese COPCs in groundwater at PI-1, pesticide and manganese data from five monitoring wells at sites near the Blackstone Airport within Fort Pickett that are considered background (WESTON 1999b) were evaluated and are discussed in Section 10.2. The four pesticide COPCs, aldrin, alpha-HCH, heptachlor, and heptachlor epoxide, were not detected in these offsite wells; however, 4,4'-DDD, alpha-chlordane, dieldrin, and endrin, were detected at similar concentrations to those at the site. Dieldrin is the major metabolite of aldrin (Montgomery and Welkom 1990). Although dieldrin was a pesticide in use in the U.S. before 1971 (Briggs 1992), its presence may reflect a widespread historical use of aldrin at Fort Pickett, which may explain the presence of aldrin at PI-1. The pesticides aldrin, alpha-HCH, heptachlor, and heptachlor epoxide were available for use in the U.S. until the early 1980's. Likewise, the trace levels of other pesticides in groundwater at PI-1 and other areas of Fort Pickett is likely due to the historic use of pesticides on the post.

In addition, the elevated manganese levels in the offsite wells (Table 8-1), appears to be reflective of the natural and acidic groundwater quality in the region. The offsite total concentrations ranged from 171 to 894 µg/L. The manganese concentrations at PI-1 ranged from 46.9 to 495 µg/L, filtered and non-filtered. Therefore, manganese in groundwater at PI-1 is most likely a natural condition of the overburden-bedrock aquifer in the area.

6.8 UNCERTAINTY ASSESSMENT

There are numerous uncertainties involved in the human health risk assessment process. These are discussed briefly in the following sections.

6.8.1 Sampling and Analysis Uncertainties

The sampling plan can have a significant impact on the results obtained in calculating human health risks at a site. To the extent that samples are taken in areas that are expected to be contaminated (biased sampling), the EPC used in calculating risk exposures and risks is likely to overestimate the actual concentration encountered at the site from random exposure across the site. This sampling bias will generally result in an overestimate of exposures and risks at a site. The soil sampling at Site PI-1 incorporated a combination of random and biased samples. As the majority of soil samples collected at Site PI-1 are biased toward potentially contaminated areas, the measured concentrations and calculated health risks would tend to be overestimated.

Some analytes that were reported as non-detects did have reporting limits above the screening criteria. These analytes are presented in Table 4-9. In surface and subsurface soil, n-nitrosodi-n-propylamine had a reporting limit greater than the RBC. This compound is reported to be a contaminant in the pesticide EPTC. However, possibility that this compound exists at PI-1 is considered negligible. This is because EPTC was not reported to be used at Fort Pickett, other detected pesticide concentrations were one to two order of magnitude less than the detection limit of this SVOC, and this SVOC was not detected in nine surface and 16 subsurface soil samples and duplicates. However, if this analyte was actually present in soil above the RBC, the risk assessment would present an underestimate of risks at the site, in that this analyte was not assessed as a COPC.

The detection limits for 10 VOCs, 12 SVOCs, and 2 metals were above the RBC screening criteria in groundwater. The VOCs included a series of chlorinated hydrocarbons, which may have been used at the site. However, these chlorinated compounds were not detected in soil or groundwater samples at the site, except for TCE detected below the RBC in shallow soil samples. The possibility of these VOCs present in groundwater is considered negligible.

The SVOCs shown on Table 4-9 are not likely present in groundwater because soil samples revealed levels two to three orders of magnitude below the SSLs. As a result, the presence of these compounds in groundwater is considered negligible. In addition, the RBCs were below detection limits for PCBs, arsenic, and thallium. It is possible that arsenic and thallium may be present in groundwater considering their detection in soil at the site. PCBs were not detected in the surface and subsurface soil samples and are not likely present in groundwater. If these analytes were actually present in groundwater above the RBCs, the risk assessment would present an underestimate of risks at the site, in that these analytes were not assessed as COPCs.

6.8.2 Uncertainties Analysis of Exposure Assessment

An analysis of uncertainties is an important aspect of the exposure assessment. It provides the risk assessor and reviewer with information relevant to the individual uncertainties associated with exposure factor assumptions and their potential impact on the final assessment.

6.8.2.1 Exposure Point Concentrations

A significant uncertainty exists with the basic approach used in arriving at EPCs for the COPCs in surface soils.

Uncertainty results from the use of $\frac{1}{2}$ detection limit for all non-detects. An objective of the guidance is to include some quantitative value for COPCs when analytical data indicate that those COPCs were not detected, so that an estimated potential intake and resultant potential risk can be calculated. This approach is referenced in Risk Assessment Guidance for Superfund (Volume 1), Human Health Evaluation Manual (EPA 1989b). However, this approach generally overestimates the average value, and results in overestimates of intakes and subsequent risks, particularly for COPCs with low frequencies of detection.

6.8.2.2 Soil Ingestion Rate

Soil ingestion rates for construction workers are based on studies performed by Hawley (1985).

6.8.3 Uncertainties of Toxicity Assessment

There are numerous uncertainties associated with the toxicity assessment. These are generally due to the unavailability of data to thoroughly calculate the toxicity of COPCs. These are described in more detail in the following sections.

6.8.3.1 Uncertainties Associated with Non-carcinogenic Effects

Interspecies Extrapolation - The majority of toxicological information comes from experiments with laboratory animals. Experimental animal data have been relied on by regulatory agencies to assess the hazards of human chemical exposures. Interspecies differences in chemical absorption, metabolism, excretion, and toxic response are not well understood, therefore conservative assumptions are applied to animal data when extrapolating to humans. These probably result in an overestimation of toxicity.

Intraspecies Extrapolation - Differences in individual human susceptibilities to the effects of chemical exposures may be caused by such variables as genetic factors (e.g., glucose-6-phosphate dehydrogenase deficiency), lifestyle (e.g., cigarette smoking and alcohol consumption), age, hormonal status (e.g., pregnancy), and disease. To take into account the diversity of human populations and their differing susceptibilities to chemically induced injury or disease, a safety factor is used. U.S. EPA uses a factor between 1 and 10. This uncertainty may lead to overestimates of human health effects at given doses.



Exposure Routes - When experimental data available on one route of administration are different from the actual route of exposure that is of interest, route-to-route extrapolation must be performed before the risk can be assessed. Several criteria must be satisfied before route-to-route extrapolation can be undertaken. The most critical assumption is that a chemical injures the same organ(s) regardless of route, even though the injury can vary in degree. Another assumption is that the behavior of a substance in the body is similar by all routes of contact. This may not be the case when, for example, materials absorbed via the gastrointestinal tract pass through the liver prior to reaching the systemic circulation, whereas by inhalation the same chemical will reach other organs before the liver. However, when data are limited these extrapolations are made, and may result in overestimates of human toxicity.

6.8.3.2 Uncertainties Associated with Carcinogenic Effects

Interspecies Extrapolation - The majority of toxicological information for carcinogenic assessments comes from experiments with laboratory animals. There is uncertainty about whether animal carcinogens are also carcinogenic in humans. While many chemical substances are carcinogenic in one or more animal species, only a very small number of chemical substances are known to be human carcinogens. The fact that some chemicals are carcinogenic in some animal species but not in others raises the possibility that not all animal carcinogens are human carcinogens. Regulatory agencies assume that humans are as sensitive to carcinogens as the most sensitive animal species. This policy decision, designed to prevent underestimation of risk, introduces the potential to overestimate carcinogenic risk.

High-Dose to Low-Dose Extrapolation - Typical cancer bioassays provide limited low-dose data on responses in experimental animals for chemicals being assessed for carcinogenic or chronic effects. The usual dose regime involves three dose groups per assay. The first dose group is given the highest dose that can be tolerated, the second is exposed to one-half that dose, and the third group is unexposed (control group) (NRC 1983). Because this dosing method does not reflect how animals would react to much lower doses of a chemical, a dose-response assessment normally requires extrapolation from high to low doses using mathematical modeling that incorporates to varying degrees information about physiologic processes in the body (NRC 1983).

A central problem with the low-dose extrapolation models is that they all too often fit the data from animal bioassays equally well, and it is not possible to determine their validity based on goodness of fit. Several models may fit experimental data equally well, but they may not all be

equally plausible biologically. The dose-response curves derived from different models diverge substantially in the dose range of interest (NRC 1983). Therefore, low-dose extrapolation is more than a curve-fitting process, and considerations of biological plausibility of the models must be taken into account before choosing the best model for a particular set of data.

6.8.3.3 Uncertainties in Toxicity Values for Specific Chemicals

There is uncertainty associated with the toxicity assessments and derivation of toxicity values for aldrin, alpha HCH and dioxin. Although there is a lack of conclusive evidence to classify these chemicals as class B, probable human carcinogen, U.S. EPA classifies them as carcinogens, B2 (EPA 1999b). Therefore, following the IRIS database these chemicals are assessed as carcinogens in the risk assessment although there is an absence of conclusive human data. This is an assumption that tends toward the conservative, biasing risks high.

6.8.4 Uncertainties in Risk Characterization

Uncertainties in the risk characterization can stem from the inherent uncertainties in the data evaluation, the exposure assessment process, including any modeling of exposure point concentrations in secondary media from primary media, and the toxicity assessment process. The individual uncertainties in these respective processes were addressed previously, in the previous sections.

Uncertainties associated with the probability of adverse impacts to human health can also be evaluated by examining the relative risk estimated for AE and upper bound reasonable RME scenarios. This type of simple probability analysis is often useful to risk managers who must balance baseline risk estimates with the expected costs and benefits of remedial activities.

6.9 RISK ASSESSMENT SUMMARY AND CONCLUSIONS

The risk assessment was conducted to assess potential non-carcinogenic effects and cancer risks from current and future site exposure.

Non-carcinogenic risks were only calculated above the EPA threshold of 1.0 for child residents. Based on a breakdown by target organ the only risk driver was manganese (target organ is the central nervous system), which had a cumulative HI of 1.4 across soil and groundwater pathways. In soil, the target organ specific HQ for the central nervous system (based on manganese) was less than 1.0, indicating no concern for adverse health risks associated with

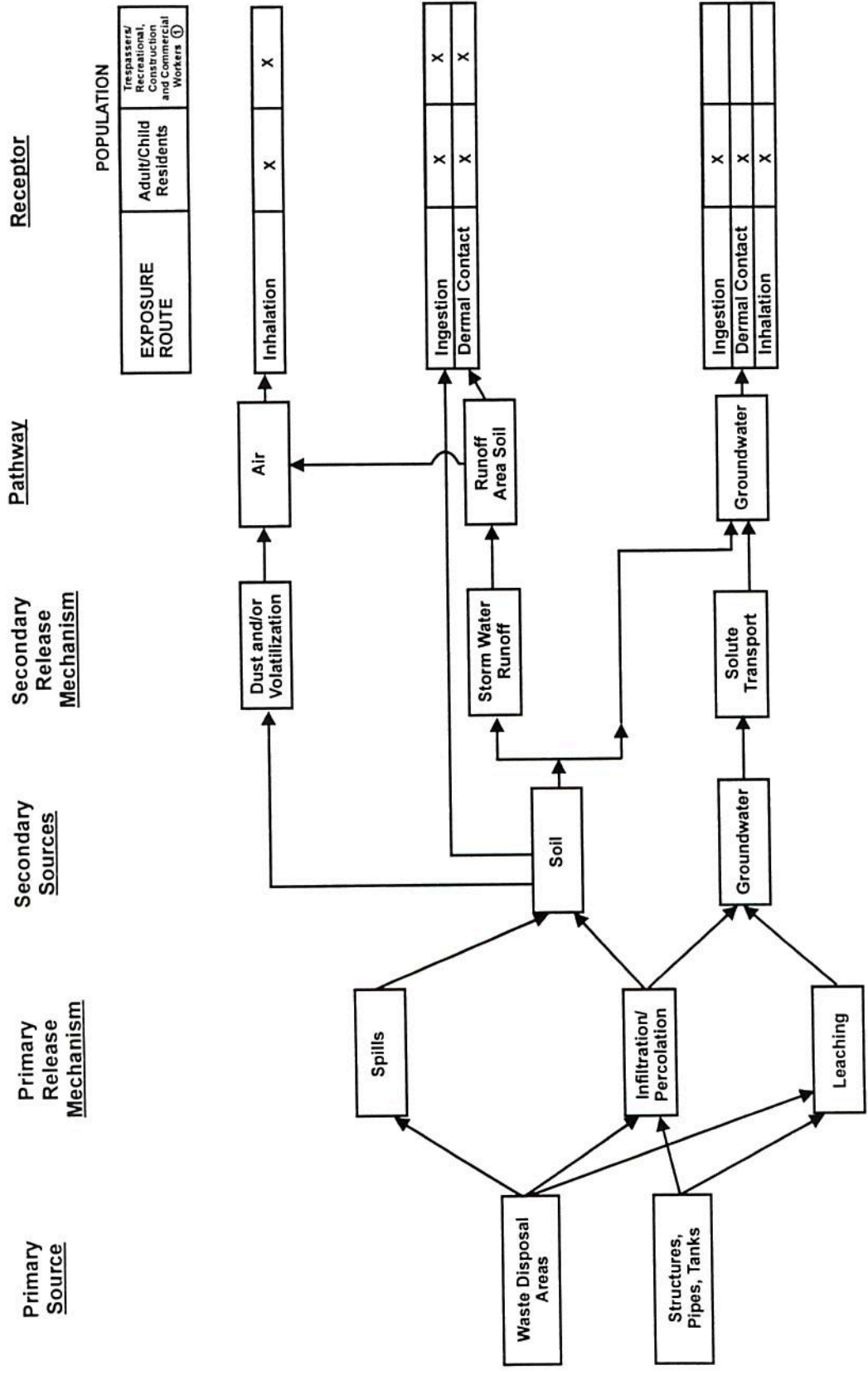
manganese in soil. Based on Fort Pickett background data, manganese detected at the site in soils is consistent with background. A post-wide background study is currently planned for Fort Pickett since background groundwater data is not available. Previous groundwater data from other BRAC sites on the post illustrate prevalent manganese concentrations. However, manganese in groundwater at PI-1 cannot be attributed to background due to the lack of adequate groundwater background data. This is discussed further in Section 10.2. Therefore, there were no unacceptable non-carcinogenic risks from Site PI-1.

Cancer risks for the site did not exceed the U.S. EPA's risk range of 10^{-6} to 10^{-4} for any receptor. Ingestion of total soil, dermal contact with total soil, and ingestion of groundwater for the resident (adult and child), ingestion of surface soil for the adolescent trespasser, and ingestion of and dermal contact with surface soil for the commercial workers were the only exposure pathways for which potential cancer risks were within the 10^{-6} to 10^{-4} range. All other exposure scenarios fell below this range. Arsenic and dioxin in soils and aldrin, alpha HCH, heptachlor, and heptachlor epoxide in groundwater were the only chemicals exceeding 10^{-6} .

Arsenic in soil exceeded 10^{-6} ; however, site arsenic concentrations were well within the background data range for Fort Pickett. Arsenic was not a COPC in groundwater. Therefore, risks calculated from arsenic in soil are attributable to background are not necessarily site related.

A consideration of Virginia DEQ acceptable cancer risk policy to the results of this risk assessment finds that although some individual chemicals exceed the goal of 10^{-6} , none of the receptors has an unacceptable cumulative cancer risk (above 10^{-4}).

Therefore the only COPCs with risks within U.S. EPA's risk targets of 10^{-4} to 10^{-6} for carcinogens and HI of 1.0 for non-carcinogens which are site-related are dioxins in soils and aldrin, alpha HCH, heptachlor, and heptachlor epoxide in groundwater. None of these exceed the risk targets. The only potential receptor populations for which these COPCs contribute risks within these risk targets are potential future resident children and adults, potential adolescent trespassers, and potential future commercial workers.



① Trespassers/recreational, and commercial worker exposure to surface soil media, not total soil.

EA EA ENGINEERING, SCIENCE, AND TECHNOLOGY INC	REMEDIAL INVESTIGATION FORT PICKETT, VIRGINIA	HUMAN HEALTH CONCEPTUAL SITE MODEL PI-1	DESIGNED BY TJP	DRAWN BY DCD	DATE 7-14-99	PROJECT NO. 61144.03
			CHECKED BY JPC	PROJECT MGR. JPC	SCALE NO SCALE	FIGURE 6-1

TABLE 6-1 SELECTION OF EXPOSURE PATHWAYS, FORT PICKETT PI-1

Scenario Timeframe	Medium	Exposure Medium	Exposure Point	Receptor Population	Receptor Age	Exposure Route	On-Site/ Off-Site	Type of Analysis	Rationale for Selection or Exclusion of Exposure Pathway
Current	Surface Soil	Surface Soil	PI-1	Tres/Recreat	Adult	Ingestion Dermal	On-Site	Quant	Trespassers may be exposed to surface soil
					Adolesc	Ingestion Dermal	On-Site	Quant	Trespassers may be exposed to surface soil
	Subsurface Soil	Air	PI-1	Tres/Recreat	Adult	Inhal VOCs	On-Site	Quant	Trespassers may be exposed to surface soil
					Adolesc	Inhal VOCs	On-Site	Quant	Trespassers may inhale VOCs from subsurface soil
	Surface Water	Surface Water	Site Surface Water	Tres/Recreat	Adult	Ingestion Dermal	On-Site	Quant	Trespassers may inhale VOCs from subsurface soil
					Adolesc	Ingestion Dermal	On-Site	None	There is no surface water found at the site
						Ingestion Dermal	On-Site	None	There is no surface water found at the site
						Ingestion Dermal	On-Site	None	There is no surface water found at the site
	Surface Soil	Surface Soil	PI-1	Comm Worker	Adult	Ingestion Dermal	On-Site	Quant	Future Commercial Workers may be exposed to surface soil
						Inhal Dust	On-Site	Quant	Future Commercial Workers may be exposed to surface soil
Future	Subsurface Soil	Air	PI-1	Comm Worker	Adult	Inhal VOCs	On-Site	Quant	Future Commercial Workers may inhale dust from surface soil
				Constr Worker	Adult	Inhal VOCs	On-Site	Quant	Future Commercial Workers may inhale VOCs from subsurface soil
				Resident	Adult	Inhal VOCs	On-Site	Quant	Future Construction Workers may inhale VOCs from subsurface soil
				Constr Worker	Child	Inhal VOCs	On-Site	Quant	Future Residents may inhale VOCs from subsurface soil
					Adult	Inhal VOCs	On-Site	Quant	Future Residents may inhale VOCs from subsurface soil
	Total Soil	Total Soil	PI-1			Ingestion Dermal	On-Site	Quant	Future Construction Workers may be exposed to total soil
				Resident	Adult	Ingestion Dermal	On-Site	Quant	Future Construction Workers may be exposed to total soil
					Child	Ingestion Dermal	On-Site	Quant	Future Residents may be exposed to total soil
				Constr Worker	Adult	Inhal Dust	On-Site	Quant	Future Residents may be exposed to total soil
		Air	PI-1	Resident	Adult	Inhal Dust	On-Site	Quant	Future Construction Workers may inhale dust from total soil
				Resident	Child	Inhal Dust	On-Site	Quant	Future Residents may inhale dust from total soil

TABLE 6-1 (CONT'D) SELECTION OF EXPOSURE PATHWAYS, FORT PICKETT PI-1

Scenario Timeframe	Medium	Exposure Medium	Exposure Point	Receptor Population	Receptor Age	Exposure Route	On-Site/ Off-Site	Type of Analysis	Rationale for Selection or Exclusion of Exposure Pathway
Future	Groundwater	Groundwater	Tap Water	Resident	Adult	Ingestion	On-Site	Quant	Future Residents may ingest groundwater as drinking water
					Child	Dermal	On-Site	Quant	Future Resident Adults may be exposed while showering
						Ingestion	On-Site	Quant	Future Residents may ingest groundwater as drinking water
						Dermal	On-Site	Quant	Future Resident Children may be exposed while bathing
	Surface Water	Air	Shower Head Bath Tub Site Surface Water	Constr Worker Resident Resident Constr Worker Comm Worker	Adult	Ingestion	On-Site	None	Residential risks are run as worst case scenario for future use
					Adult	Inhal VOCs	On-Site	Quant	Future Residents may inhale VOCs while showering
					Child	Inhal VOCs	On-Site	None	VOCs inhaled while bathing are minimal
					Adult	Ingestion	On-Site	None	There is no surface water found at the site
						Dermal	On-Site	None	There is no surface water found at the site
					Child	Ingestion	On-Site	None	There is no surface water found at the site
						Dermal	On-Site	None	There is no surface water found at the site
					Adult	Ingestion	On-Site	None	There is no surface water found at the site
						Dermal	On-Site	None	There is no surface water found at the site
					Adult	Ingestion	On-Site	None	There is no surface water found at the site
						Dermal	On-Site	None	There is no surface water found at the site

7. ECOLOGICAL RISK SCREENING

This chapter presents the purpose, rationale, and methods used for the evaluation of ecological risks at PI-1. A Tier I Ecological Risk Assessment (ERA) is a preliminary, initial screening process designed to estimate the likelihood of ecological risk, and to provide a basis for the necessity of a more thorough Tier II ERA. The decision to proceed to a Tier II ERA is made as a part of the risk management decisions, specifically using Scientific Management Decision Points (SMDP) built into the EPA ERA Process (EPA 1997c).

7.1 OBJECTIVES

The approach undertaken, as outlined in the Final RI/FS Work Plan (EA 1999), incorporates the latest available guidance and concepts on ecological risk assessment:

- *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA 1997c), and
- *Guidelines for Ecological Risk Assessment* (EPA 1998b),
- *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (EPA 1999c), and
- *Tri-Service Procedural Guidelines for Ecological Risk Assessments* (U.S. Army Chemical Research 1996).

The overall objectives of the ecological risk screening approach are to identify the ecological COPC and to assess potential risks to the environment. This approach will allow the Remedial Project Manager (RPM) to make informed decisions regarding environmental protection and regulatory compliance.

The screening level assessment comprises the first two steps of an eight-step process of ecological risk assessment at Superfund sites, or sites otherwise required to follow the CERCLA process. The screening level process, as applied to the site, consists of 2 steps:

- 1) Problem Formulation and Ecological Effects Evaluation;
- 2) Exposure Estimate and Risk Calculation.

The screening level assessment approach corresponds to the Problem Formulation box in the top part of Figure 7-1 and Steps 1 and 2 in Figure 7-2.

7.2 PROBLEM FORMULATION AND ECOLOGICAL EFFECTS EVALUATION

Problem formulation represents the scoping stage of all ecological risk assessments. Existing information is examined, the site visited, receptors of concern identified, a conceptual model for the site is developed to identify potential exposure pathways, and preliminary assessment and measurement endpoints are identified. Ultimately, problem formulation generates one or more questions, speculations, or hypotheses regarding current or future man-induced changes to the environment. These questions are answered or hypotheses tested by collecting information during the analysis phase. The ecological significance of the results are evaluated during risk characterization.

Problem Formulation and Ecological Effects Evaluation consists of several elements:

- Environmental Setting and Contaminants at the Site
- Contaminant Fate and Transport
- Selection of Receptors of Concern
- Development of a Conceptual Site Model
- Assignment of Assessment and Measurement Endpoints

7.2.1 Environmental Setting and Contaminants at the Site

According to EPA (1997c) to begin the screening level problem formulation, there must be at least a rudimentary knowledge of the environmental setting and chemical contamination at the site.

7.2.1.1 Environmental Setting

PI-1, a former vehicle maintenance area, is approximately one acre in size and is located at the north end of Fort Pickett, adjacent to Business Route 460 (Figure 1-2). A site visit was conducted by EA biologists on 18 January 2000 to evaluate the environmental setting of PI-1. This section presents the results of that site visit. Table 7-1 presents an ecological inventory for PI-1.

The site consists of an open field habitat with small areas of mixed coniferous and deciduous forest types. A field adjacent to the site was recently used for row cropping. The only structure on the site was a concrete slab once used for equipment maintenance.

Two small areas of woods were present; both close to Route 460. The wooded stand on the western end of the site was the larger of the two stands and included loblolly pine (*Pinus taeda*), eastern red cedar (*Juniperus virginiana*), red maple (*Acer rubrum*), sweet gum (*Liquidambar styraciflua*), white oak (*Quercus alba*), southern red oak (*Quercus falcata*), turkey-foot oak (*Quercus laevis*), black oak (*Quercus velutina*), American beech (*Fagus grandifolia*), sweetbay magnolia (*Magnolia virginiana*), alder (*Alnus* sp.), Christmas fern (*Polystichum acrostichoides*), trout lily (*Erythronium americanum*) and greenbrier (*Smilax rotundifolia*). The understory of this wooded area was generally thin and easy to walk through. Very little herbaceous vegetation was observed at ground level and the forest floor had a thick layer of leaves and pine needles present. This area contained drainage paths leading from the cultivated field to a culvert at the base of Route 460.

The smaller wooded stand located in the eastern end of the site was not as diverse as the larger stand and included loblolly pine as the dominant canopy species with few sapling-sized American beech and eastern red cedar present. The understory present in this area was slightly thicker than the wooded area to the west and included a greater density of sapling-sized individuals. *Yucca* (*Yucca filamentosa*) was occasionally present within this wooded area and probably escaped cultivation from a nearby homestead.

The bird species observed or heard on the site included blue jays (*Cyanocitta cristata*), slate-colored juncos (*Junco hyemalis*), American crows (*Corvus brachyrhynchos*), and killdeer (*Charadrius vociferus*). One warbler nest was viewed in a patch of greenbrier along with a large bird or squirrel nest 20 feet high in a cedar tree.

The forest stand on the western side of the site had a small, ephemeral drainage channel that proceeded under Route 460 and continued through a mowed-maintained field and into another forest stand off-site. The channel became larger as it progressed through the woods on the northern side of Route 460. Several small hole openings were observed around this channel indicating possible shrew (*Blarina brevicauda*) or vole (*Microtus pennsylvanicus*) activity, since

shrews and voles prefer moist low areas. This water area may also be used by frogs during the breeding season.

Deer (*Odocoileus virginianus*) tracks were observed within the cultivated field adjacent to PI-1. Little evidence of deer browsing was observed within the forest probably due to the close proximity to the road. The habitat present at PI-1 is good for edge species such as rabbits (*Sylvilagus floridanus*).

7.2.1.2 Contaminants Previously Detected at the Site

PI-1 was formerly used as a vehicle maintenance area and service station. Vehicles, including buses, were frequently parked onsite for maintenance. Observations of past photographs (1949 – 1959) identified areas with stains and disturbed ground in the southern portion of the site where vehicles, mechanical parts, and waste oil containers were possibly stored. Staining was also observed on the photographs near one of the former buildings. A site reconnaissance conducted in October 1997 by Roy F. Weston, Inc. as part of the PA/SI revealed no signs of staining, but stressed vegetation was observed on the west side of the larger building slab (WESTON 1998).

For the PA/SI, WESTON (1998) completed three soil borings at suspect locations around the large building slab. The analytical results of the surface soil of this investigation detected one pesticide (alpha-chlordane), two volatile organic compounds (trichloroethene [TCE] and methylene chloride), one SVOC (di-n-butylphthalate), and seven PAHs in the surface soil samples. The concentrations of these organic compounds did not exceed the ecological screening criteria.

The concentrations of eleven metals (aluminum, arsenic, beryllium, chromium, copper, mercury, nickel, lead, selenium, vanadium, and zinc) exceeded the ecological screening criteria at PI-1. These metals were distributed throughout all three soil borings.

7.2.2 Contaminant Fate and Transport

As discussed in Chapter 5, potential migration pathways at PI-1 include surface water, sediment transport, groundwater flow, and air transport. Subsurface and surface soil, although not necessarily transport media, may be considered a source, if sufficiently contaminated, that can affect all of these pathways.

The transport of metals via the surface water pathway may be a factor at this site. Since metals can adhere to soil/sediment via physical adsorption, oxidation to hydrous oxide compounds, and formation of insoluble organic complexes, transport of adhered metal COPC may occur during high surface runoff events. Sediment transport is governed by surface water runoff and surface cover and may also be a migration route for metals and also organics. The potential for contaminants in groundwater to migrate offsite and the potential for risk cannot be discounted. Air transport was not considered a significant pathway at PI-1. Metals are of concern because of their toxicity and the fact that some metals can methylate and become more toxic to ecological receptors.

7.2.3 Selection of Receptors of Concern

Ecological receptors of concern (ROC) are species or guilds of species that are important to the ecology of the site and which may be susceptible to chemical constituents released at the site. ROC examples could include an area of riparian wetland, a particular bird species, a benthic community, or a fish. Selection of ROC is systematic, representative, and ecologically-based to ensure that assessment endpoints (see Section 7.2.5) are adequately addressed. Criteria used to identify ecological ROC include the following:

- Presence – known or expected to occur onsite;
- Susceptibility – exposure pathway is likely complete and of sufficient duration/magnitude;
- Representative – of the food web and/or guild;
- Data Availability – sufficient and appropriate type of toxicity and exposure information; and
- Societal Importance – species merits public attention.

In some instances, the selected ROC represents an ecological guild (a group of species using similar resources such as food or location in a similar manner).

Ecological ROC can be classified into three broad categories: (1) ecologically important, (2) recreational or commercially important, and (3) threatened and endangered species. Ecologically important ROC substantially contribute to the structure (numbers and biomass) and function (energy flow and nutrient cycling) of the site's ecosystem. This may include primary producers, primary, secondary, and tertiary consumers, as well as their respective food base. Primary producers are represented by plants, which take energy from sunlight and nutrients from soil pore water. Primary consumers represent the first link of a food web and are represented by soil invertebrates. Secondary consumers consume the primary consumers, and are eaten by tertiary consumers. A bird such as the American Crow is an example of a secondary consumer. Tertiary consumers represent the top or highest level of the food web and are primary candidates for ROC selection when persistent bioaccumulative compounds such as polychlorinated biphenyls (PCB) and dichloro-diphenyl-trichloroethane (DDT) are COPC. This is because bioaccumulative chemicals tend to bio-magnify as they are transferred through the food web to higher consumer levels. For this type of COPC, contaminant concentrations are lowest in primary consumers and highest in tertiary consumers, and may thus represent risk to the tertiary consumer or to an organism consuming this tertiary consumer (such as a fox).

In the following sections, potential ROC for Site PI-1 are discussed in the context of endangered and non-endangered species. In Section 7.2.3.3, receptors of concern are designated for Site PI-1, and rationales for their selection are provided.

7.2.3.1 Threatened or Endangered Species

Appropriate Federal and State Agencies were contacted to determine if Site PI-1 is habitat for any threatened or endangered species. Responses from the Agencies are shown in Appendix I, and note that no threatened or endangered species are found, nor are any threatened or endangered species expected to be found at the site.

7.2.3.2 Non-Threatened and Non-Endangered Species

The terrestrial plant community at Site PI-1 is a mixture of native and introduced plant species. These plant communities are representative of a previously disturbed community. The mixed

coniferous and deciduous forest areas found at PI-1 represent native growth and the open field habitat is the result of cutting that took place some time in the past. The plant communities of Site PI-1 were described in Section 7.2.1.1. Site PI-1 has a variety of vegetation including a mixture of open fields and two small areas of forest habitat. Dominant trees included a mix of pines and hardwoods. Pines included loblolly and eastern red cedar and hardwoods included maples, sweet gum, and oak. Terrestrial vegetation is considered a potential ecological receptor at PI-1 because of its critical role as the primary producers for the site.

Because of the time of year of the site visit (January), there was no evidence of insects at PI-1. However, soil invertebrates are very important to the fertility of soils because they influence soil turnover, mineralization, humidification, soil porosity, aeration, water infiltration, and soil-water retention. Because of their close association with soil and ecological importance in the maintenance of soil fertility, earthworms and other soil invertebrates are considered ecological receptors of concern at PI-1.

A number of birds were directly observed during the January 2000 ecological site visit including the American crow, blue jays, juncos, and killdeer. In addition, the habitat would indicate that many other birds could inhabit the area such as cardinals, starlings, and woodpeckers (Section 7.2.1.1). Two birds will be used as guild representatives for birds. The American kestrel is representative of raptors potentially found at the area, and the American robin is representative of thrushes in the area.

Numerous small and medium mammals may use PI-1 as habitat. Evidence of potential shrew or vole were observed as well as deer tracks. As noted during the site visit, PI-1 represents good habitat for edge species such as rabbits. Consequently, the short-tailed shrew, meadow vole, eastern cottontail and the red fox have been selected as mammalian receptors of concern.

7.2.3.3 Designated ROC for PI-1

PI-1 is small, containing approximately one acre and is representative of “edge” habitat, representing the border between open fields and forested areas. There are no permanent surface water bodies on or near PI-1, therefore all ROC are terrestrial. The ROC selected for this site include receptors that would utilize the habitat at PI-1. The following representative species were designated as ecological receptors of concern for Site PI-1.

Terrestrial Plants - Plant communities are important to the structure and function of terrestrial ecosystems and are intimately associated with soils on a site. No observations of endangered plants were made on the site, but other native plants exist at PI-1. Plants represent the base of food webs, and thus are critical receptors of concern.

Terrestrial Invertebrates (Earthworms) – No invertebrates or insects were observed at PI-1 due to the time of year in which the site visit was made. However, soil invertebrates are very important to the fertility of soils because of their role in the aeration and turnover of surface soils. Earthworms will serve as a guild representative for the other invertebrates (i.e., beetles, termites, grasshoppers, butterflies, spiders, moths, and wasps). Earthworms are in continuous contact with any soil associated contaminants that may be present.

Avian Species – Numerous avian species may utilize the habitat at PI-1. Two avian receptors of concern have been selected for PI-1, the American robin and kestrel. The robin is a thrush, and consumes relatively large amounts of invertebrates and fruit. The American kestrel is a small raptor that utilizes edge habitat such as that found at PI-1.

Mammalian Species – Small and medium mammals utilize edge habitat similar to PI-1. Four mammal species have been selected as ROC, all of which have available exposure and toxicological information. Both the short-tailed shrew and meadow vole are small mammals, however, they consume different food. The short-tailed shrew is a carnivore that consumes almost its body weight per day in food. The meadow vole is a herbivore, and does not consume as much food per day. The medium mammal ROC is the eastern cottontail, which is a herbivore that prefers the edge habitat represented by PI-1. The red fox has been selected as a large carnivorous mammal ROC.

7.2.4 Ecological Risk Conceptual Site Model (CSM)

The CSM is an end product of Problem Formulation. It contains a description of the physical and ecological characteristics of the site, potential exposure scenarios, ROC, and assessment and measurement endpoints.

A major element in every CSM is a description of the exposure scenarios. This consists of four elements.

- Source of COPC release and release mechanism(s);
- Transport medium and mechanism of transfer from primary to subsequent media;
- Point (or area) of potential ROC contact with the COPC; and
- Route of uptake by the ROC (bioconcentration, ingestion of soil or food)

Potential sources include past activities associated with PI-1. The site was previously used as a maintenance facility for vehicles that resulted in potential releases to soil in the area. Surface soil is an exposure media for terrestrial receptors. There could be movement by constituent releases by infiltration to the subsurface soils and to groundwater, however there are no complete pathways for ecological receptors to subsurface soil or groundwater.

COPC sequestered in secondary source material may be released via several mechanisms, including incorporation into the food web. Terrestrial receptors may directly contact or ingest surface soil at the site. Through the process of trophic transfer, or, in the case of bioaccumulative COPC, trophic magnification, biota can serve as vectors for COPC transport and expose higher level animals if ingested by ROC.

The CSM is based on simple direct contact and terrestrial food-web models (Figure 7-3). There are no permanent water bodies at this site; therefore the secondary source of COPC exposure is surface soil. This exposure may be effected through direct contact with or ingestion of surface soil, or by ingestion of plant or animal tissue that had been exposed via surface soil. Exposure pathways and routes include:

- Direct Contact with Surface Soil - This exposure route is important for uptake of COPC for plants and for soil invertebrates. Most vertebrates when foraging may have the potential to be exposed to COPC via dermal contact. Incidental surface contamination of scales, feathers, or hair that is subsequently ingested during grooming is accounted for in the incidental soil ingestion pathway.
- Ingestion of Food (i.e., plants and biota that have taken up constituents from soil) - Herbivores and predators that forage in the terrestrial habitats may ingest plants or animal prey that have bioaccumulated COPC from surface soils.
- Incidental Ingestion of Surface Soils - Herbivores and predators that forage in the terrestrial habitats may incidentally ingest some surface soil with their food or during

other activities such as grooming. Soil invertebrates ingest surface soil and leaf litter during feeding, but it is difficult to distinguish between uptake as a result of direct contact with surface soils and that as a result of ingestion of surface soils because of their intimate association with surface soils.

On the basis of this evaluation, there are complete exposure pathways for surface soil in terrestrial habitats at PI-1. From this environmental medium, some COPC could bioaccumulate in plants and prey animals that may be eaten by other consumers.

7.2.5 Assessment and Measurement Endpoints

EPA (1998b) guidance stresses the importance of ecologically significant endpoints. The failure to select such an effect for evaluation brings little value to the decision-making process. Several criteria are applicable for assessment endpoint selection (Suter, 1993; EPA, 1998b):

- Unambiguous Definition – assessment endpoints should indicate a subject and a characteristic of the subject (such as American robin reproduction).
- Accessibility to Prediction and Measurement – assessment endpoints should be reliably predictable from measurements.
- Susceptibility to the Hazardous Agent/Stressor – Susceptibility of an organism (plant or animal) results from the combination of potential for exposure and the sensitivity to the concentrations of contaminants or other stressors of concern.
- Biological Relevance – Biological relevance of impacts to an individual organism is determined by the importance of the impact to higher levels of biological organization such as populations or communities.
- Social Relevance and Policy Goals – Assessment endpoints should be of value to decision-makers and the public. The assessment endpoints should represent an effect that would warrant consideration of site remediation or alteration of project plans. Assessment endpoint selection should also include endpoints that may be mandated legally (e.g. protected species).

The extent to which these items are considered will vary from site to site, and are dependent on several factors including the level of public involvement, the ecological character of the site, and the lead regulatory agency involved in the assessment.

The selection of assessment endpoints must be based on the fundamental knowledge of the local ecology. Assessment endpoints typically relate to an effect on a population or community. Survival of a particular species of earthworm is an example of a population level assessment endpoint. Community level assessment endpoints could include the primary productivity of the edge community found at PI-1. Examples of endpoints representing guilds of species are useful in that they convey information beyond the indicator species identified in the endpoint itself. While beyond the scope of an ecological screening, an assessment endpoint involving a community index may provide more information about a site than an analysis of one species. Consequently, it is important to note that confirmation of the deleterious effects at the community level is an inherent confirmation that population level effects are occurring (Hartwell, 1997).

Based on previous activities at PI-1 and the findings of the site visits, ecological ROC may be exposed to COPC through surface soil exposure. COPC previously released into the soil at this site may be ingested via soil and food (i.e., plants and biota that have taken up constituents from soil).

Based on the ROC observed during the site visits, existing habitat, and the above observations, the following ecological assessment endpoints are defined and are shown in Table 7-2:

1. Protection of plant communities to insure that COPC in soil do not have unacceptable adverse effects on survival, growth, and reproduction of key plant species which may result in adverse effects to the community structure such as diversity or biomass. In addition, because plants are used as a food source for higher trophic level consumers, an assessment endpoint is the protection of plant communities to avoid COPC tissue levels that would lead to significant bioaccumulation by and unacceptable adverse effects to consumers.
2. Protection of soil-invertebrate communities to insure that COPC in soil do not have unacceptable adverse effects on survival, growth, and reproduction of key soil invertebrate species which may result in adverse effects to the community structure such as diversity or

biomass. Soil invertebrates are also responsible for critical soil processes such as organic decomposition and nutrient cycling.

3. Protection of birds, represented by the omnivorous American robin and carnivorous American kestrel, to insure that ingestion of COPC in food items and soil does not have unacceptable adverse impacts on survival, growth, and reproduction.
4. Protection of mammals, represented by the small carnivorous short-tailed shrew, small herbivorous meadow vole, medium sized herbivorous eastern cottontail, and large carnivorous red fox to insure that ingestion of COPC in food items and soil does not have unacceptable adverse impacts on survival, growth, and reproduction.

Measurement endpoints are measurable ecological characteristics that are related to the assessment endpoints (EPA 1998b). Because it is difficult to “measure” assessment endpoints, measurement endpoints were chosen that permit inference regarding the above-described assessment endpoints. Two measurement endpoints were selected for this risk assessment:

- Media Chemistry for Surface Soils – the measurement of chemical constituent concentrations in surface soil provides the means, when compared to appropriate background and ecotoxicological-based screening concentrations, for drawing inferences regarding the first measurement endpoint above. Because soil invertebrates and plants are in direct contact with the soil, direct measurement of soil concentrations is an appropriate endpoint.
- Calculated Dietary Doses – measurement endpoints to address the third and fourth assessment endpoints consist of calculated chemical doses in the diet. The primary exposure route for birds and mammals is via intake of food and incidental soil. The knowledge of specific COPC concentrations in surface soil cannot be used to address this assessment endpoint directly. Rather, these measurements are used in conjunction with food ingestion rate and other factors to calculate the daily intake, or dietary dose of a constituent. These are then compared to toxicological thresholds to address the assessment endpoint, as described in Section 7.6 below. Because this is a Tier I ecological screening, the conservative assumption has been made that food concentrations are equivalent to those found in the surface soil.

The above described assessment and measurement endpoints are summarized (Table 7-2) specifically to address endpoints used for PI-1.

7.3 COPC SCREEN

7.3.1 Risk Calculation

Every risk assessment begins with a list of analytes. These analytes include compounds and/or elements known or suspected to have originated from site-related activities. In this case these are metals, PAH, volatiles, and pesticides. Analytes not detected, detected infrequently, or at non-hazardous concentrations may be candidates for elimination. Analytes known or suspected to have originated from site-related activities (metals, PAH, volatiles, pesticides) remaining after the screening process are constituents of potential concern (COPC).

The screening process that identifies COPCs must be environmentally conservative. That is, the process must not eliminate analytes which could pose potential ecological risk. In statistical terms, the screening process must minimize the potential for false negatives. This potential is minimized by using conservative assumptions and appropriate screening values during the COPC screening process. If possible, these screening values should be toxicologically based.

This process is shown in Figure 7-4 and described in detail below.

7.3.1.1 Step 1 - Analysis and Reduction of Chemistry Data

Validated data were received, reviewed, reduced, and placed in a discrete database. Data reduction followed procedures outlined in the RI/FS Work Plan (EA 1999).

7.3.1.2 Step 2 - Identification of Screening Values

On a national basis, EPA has not recommended any soil screening values. The primary reason given is that "Methods to address toxicity in soils have not been sufficiently developed..." (EPA 1996a). Therefore, screening values recommended for soil were taken from EPA Region III BTAG (EPA 1995a) and Oak Ridge (Efroymson et al. 1997 a,b). The lowest or more valid values from these three sources were selected as the ecological soil screening criteria. The screening values are presented on Table 7-3.

7.3.1.3 Step 3 - COPC Identification/Screening Index

The maximum site concentration for each medium was compared to the corresponding screening values. The comparison was done by dividing the site maximum by the screening value to produce a Screening Quotient (SQ). The SQ is a unitless ratio that reflects the relationship of the site concentration to the screening value. If the site maximum was less than the screening value ($SQ < 1.0$), that analyte was eliminated. If the site maximum exceeded the screening value ($SQ > 1.0$), that analyte was retained as a COPC. In the latter case, the SQ reflects the magnitude of exceedance of the screening value by the site concentration. Calcium, magnesium, potassium, and sodium were included in the screening tables, but were not screened because of their importance as essential nutrients.

Discussion of COPC

The occurrence, distribution, and selection of ecological COPCs in surface soil is summarized in Table 7-3 and discussed below.

Twelve metals (aluminum, antimony, beryllium, chromium, copper, lead, manganese, mercury, nickel, thallium, vanadium, and zinc) and seven PAHs (2-methylnaphthalene, benzo[b]fluoranthene, chrysene, fluoranthene, naphthalene, phenanthrene, and pyrene) were identified as COPCs in surface soil because the maximum concentrations of these analytes exceeded the respective screening values (Table 7-3). In addition, one SVOC (dibenzofuran) and one VOC (acetone) were retained as COPCs based on the absence of screening values. Screening quotients (SQ) for metals ranged from 1.3 for manganese to 11,800 for lead. Screening indices for PAHs ranged from 1.1 for fluoranthene to 6.6 for 2-methylnaphthalene.

7.4 EXPOSURE ASSESSMENT

Exposure assessment is a key component of risk quantitation, linking contaminants to receptors through complete pathways. Exposure refers to the degree of contact between ecological receptors at a site and the COPC. Based on the CSM described in Section 7.2.4, ecological receptors at Site PI-1 were assumed to be exposed to COPC in surface soil either through direct contact, or via dietary food web. In either case, the starting point for the evaluation is the concentration in the surface soil.



The relevant pathway for terrestrial plant communities is chronic exposure to surficial soil contaminants that may exhibit a detrimental effect on plant survival and growth. This exposure assessment was very conservative and was set up such that soil concentrations were compared to vegetation screening values. It is assumed that the COPC are 100% bioavailable to the plants for uptake. Risk to terrestrial plants is based on a calculation of an Ecological Quotient:

$$\text{Ecological Quotient} = \text{Maximum Surface Soil Concentration} / \text{Reference Toxicity Value}$$

The relevant pathway for terrestrial invertebrate communities is chronic exposure to surficial soil contaminants that may exhibit a detrimental effect on invertebrate survival and growth. This exposure assessment was very conservative and was set up such that soil concentrations were compared to soil invertebrate screening values. It is assumed that the COPC are 100% bioavailable to the invertebrates for uptake. Risk to terrestrial invertebrates is based on a calculation of an Ecological Quotient:

$$\text{Ecological Quotient} = \text{Maximum Surface Soil Concentration} / \text{Reference Toxicity Value}$$

The relevant pathway for terrestrial mammalian and avian ROC is chronic exposure to surficial soil contaminants due to dietary uptake. The ROC occupy different feeding guilds, but have diets that contain potential vectors for site-related soil contaminants. This exposure assessment was very conservative and was set up such that herbivore ROC (meadow vole and eastern cottontail) consumed 100 percent vegetation and omnivore/carnivore ROC (American robin, American kestrel, short-tailed shrew, red fox) consumed 100 percent of non-vegetation food (invertebrates, small mammals, etc.). Incidental ingestion also was included in this assessment.

No site-specific vegetation, invertebrate or mammal concentrations were available; therefore, COPC concentrations in food organisms were assumed to be at the same concentrations as the soil. This exposure is particularly conservative. It substitutes soil for vegetation, invertebrates or mammals that organisms would typically ingest as their main food items. In addition, it assumes that all food is on a dry-weight basis but this food is consumed at a much higher wet-weight basis; consequently, dietary doses (and risk) are overestimated.

Dietary exposures for ROC have been estimated as body-weight-normalized daily doses for comparison to a body-weight-normalized daily dose TRV. The daily dose for a given receptor to

a given COPC is given by summing the products of feeding rate and food items and multiplying the sum by the total feeding rate and a habitat usage factor (assumed to be 100 percent for this food web). Separate doses are presented for soil and food contributions, and these are summed to produce the total dose for each ROC.

$$Dose_{total} = Dose_{food} + Dose_{soil}$$

where:

- $Dose_{total}$ = Total daily dose of COPC received by receptor; mg COPC/kg-body wt./day
- $Dose_{food}$ = Daily dose of COPC received by receptor; mg COPC/kg-body wt./day from food items
- $Dose_{soil}$ = Daily dose of COPC received by receptor; mg COPC/kg-body wt./day from incidentally ingested soil

The total dose from food¹ is given by:

$$Dose_{food} = F_f \times U \times C_f$$

where:

- F_f = Total daily feeding rate in kg food/kg-body weight of ROC/day (wet basis)
- U = Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web
- C_f = Concentration of COPC in food; assumed to be the same concentration as soil (mg chemical/kg food)

The total dose from incidental soil is given by:

$$Dose_{soil} = F_s \times U \times C_s$$

¹ The reader is reminded that "food" or " $Dose_{food}$ " herein refers to soil. In this simplified, Tier 1 food web, COPC concentrations in soil are used as surrogates for concentrations in actual food items, the latter concentrations being unknown. This is distinguished from "incidental soil" or " $Dose_{soil}$ " which is the best estimate of soil that is ingested accidentally or incidentally with normal food items and during grooming.

where:

- F_s = Total daily incidental soil feeding rate in kg soil/ kg-body weight of ROC/day (wet basis)
 U = Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web
 C_s = Concentration of COPC in soil; mg chemical/kg soil (dry basis)

Lastly, the total daily soil feeding rate is given by:

$$F_s = F_f \times F_{xsoil}$$

where:

- F_s = Total daily incidental soil feeding rate in kg soil/day (wet basis)
 F_f = Total daily feeding rate in kg food/day (wet basis)
 F_{xsoil} = Fraction incidental soil ingestion as a proportion of food ingestion rate

Information necessary for calculation includes: organism body weight (BW), food ingestion rate (F_f), fraction incidental soil ingestion as a proportion of food ingestion rate (F_{xsoil}), and analyte concentrations of ingested materials. Ingested media include both abiotic (soil) and biotic (food item) materials. As discussed earlier, vegetation and animal food items were represented by the maximum concentration as found in soil (dry weight). Information specifically relevant to the ecology of the ROC (i.e., body weights, food ingestion rates, and incidental soil ingestion rates) was obtained from published sources selecting conservative (minimum body weight, maximum ingestion rate for this Step 2 ecological risk assessment (Table 7-4). The primary source used for these exposure parameters was EPA (1993b).

7.5 TOXICITY ASSESSMENT

Reference Toxicity Values used for terrestrial plant risk characterization are the toxicological benchmarks for terrestrial plants provided by Oak Ridge National Laboratory (ORNL) (Efroymson et al. 1997a). These benchmarks were derived based on growth and yield parameters which is representative of community effects rather than the effect on an individual.

Risk to soil invertebrates was determined using the earthworm as a representative species for the guild. The Reference Toxicity Values used for earthworms are from Efroymson et al. (1997b). Earthworm toxicological benchmarks were derived based on growth, reproduction, and activity parameters.

EPA (1997c) guidance specifies that a screening ecotoxicity value should be “equivalent to a documented or best conservatively estimated chronic NOAEL.” Since there is wide variation in the literature on No Observed Adverse Effect Levels (NOAELs) and since conservative estimates have been used to obtain both screening values and exposures, HQ were also calculated for conservatively estimated Lowest Observed Adverse Effect Levels (LOAELs) to provide some frame of reference for the results.

This section discusses the potential effects associated with exposure to several of the COPC, and serves as the basis for the TRVs displayed in tables 7-5 and 7-6.

Aluminum. The wildlife benchmarks for aluminum are based on two toxicity studies (Sample et al. 1996). Toxicity was noted in mice through reduced growth of second and third generations at an oral dose of 19.3 mg/kg/day in the form of aluminum chloride. Ringed dove reproduction was not significantly affected at dietary dose of up to 111.4 mg/kg/day in the form of aluminum sulfate.

Antimony. Antimony occurs in nature mainly as sulfides or oxides. Antimony may have an affinity for clay and other mineral surfaces (Bodek et al. 1988). The antimony benchmark was based on one study (Sample et al. 1996). Mice were dosed with antimony potassium tartrate at 5 mg/L oral dose. Median lifespan was reduced among female mice exposed to the 5 mg dose and the study considered exposure throughout the entire lifespan.

Chromium. Chromium is found in three major states in the environment: chromium (0), chromium (III), and chromium (VI). Chromium III occurs naturally, while the other two forms are produced by industrial processes. A study of the reproductive effects and longevity of rats exposed to trivalent chromium reported no effects at levels as high as 2,737 mg/kg/day. Likewise, no reproductive effects were observed in black ducks receiving 1 mg/kg/day of chromium over a 10-month period. These studies were used in Sample et al. (1996) in derivation of wildlife benchmarks.

Copper. Copper is an essential trace element for plants and animals but becomes toxic at concentrations only slightly higher than essential levels. Copper can be bioaccumulated by biota, but does not appear to be biomagnified. Bioaccumulation and biotransformation play an important role in the fate of copper. Copper toxicity to organisms is affected by several factors such as diet, age, and water chemistry.

Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg/day of copper as copper sulfate. Increased mortality rates in mink offspring have been observed at copper levels above 11.71 mg/kg/day. Reduced growth was observed in chicks given 46.97 mg/kg/day of copper for 10 weeks. No effect was observed at 33.21 mg/kg/day. The latter two studies were used by Sample et al. (1996) in deriving the wildlife benchmarks for copper.

Lead. Lead adversely affects survival, growth, reproduction, development, and metabolism of most species under controlled conditions, but its effects are substantially modified by numerous physical, chemical, and biological variables. Significant differences between species in response to lead poisoning exist and the effects are more pronounced with organic than with inorganic lead. Lead exposure can produce many types of adverse effects among which are modifications of the function and structure of kidneys, bones, and the central nervous system. Two studies were used by Sample et al. (1996) to derive wildlife benchmarks. Rats provided with 80 mg/kg/day in their diet produced young with significantly lower weight and kidney damage. No effects were observed at 8 mg/kg/day. American kestrels given 3.85 mg/kg/day for 7 months showed no significant effects on reproduction.

Vanadium. Vanadium is a naturally occurring element in the environment, most often in soils and rocks. When vanadium is broken down via weathering of rocks, it does not dissolve in water. This element is carried in a fashion similar in manner in which sand particles are carried. Vanadium administered to rats prior to and thorough reproduction resulted in increased pup mortality and decreased weight of offspring at concentrations as low as 2.1 mg/kg/day. Mallard ducks were administered 11.38 mg/kg/day for 12 weeks with no observed effects on mortality, body weight, or blood chemistry. These data were used by Sample et al. (1996) for deriving wildlife benchmarks.

Zinc. Zinc occurs naturally in both organic and inorganic forms. The fate of zinc in soils appears to be pH-dependent. Zinc is an essential trace element in animal nutrition. To terrestrial organisms, it is only toxic at high levels, usually from an overload of homeostatic mechanism for

absorption and excretion. Changes in blood chemistry and increased mortality to mallards was shown when they were given subchronic oral doses of 170 mg/kg body weight daily. Rat reproduction was effected following subchronic doses of 97 mg/kg/day of zinc. Rats administered 320 mg/kg/day during gestation showed increased rates of fetal resorption and reduced fetal growth rates. No effects were observed at 160 mg/kg/day. Mallard duck mortality was significantly increased at doses as low as 3000 mg/kg/day during a 60-day exposure period. The last two studies were used in the derivation of wildlife benchmarks (Sample et al. 1996).

PAHs. PAHs are moderately persistent in the environment, and may potentially cause adverse effect to vegetation and wildlife. A variety of adverse biological effects have been reported in numerous species of organisms under laboratory conditions, including carcinogenic effects, as well as effects on survival, growth, and metabolism (Eisler 1987). One study regarding PAH toxicity to terrestrial organisms was used by Sample et al. (1996) to develop wildlife benchmarks. Benzo(a) pyrene doses of 160 mg/kg/day to mice greatly reduced pregnancy rates and viability of litters. Doses as low as 10 mg/kg/day resulted in significantly reduced pup weights.

Example Food-Web Calculation

An example HQ calculation provided below estimates the potential for risk for the case where the short-tailed shrew is exposed to soil containing the maximum concentration of lead.

The maximum concentration of aluminum reported in surface soil (dry weight basis) was determined to be 118 mg/kg (Table 7-3).

The following equation provides the dose to the receptor from food ingestion:

$$\begin{aligned} Dose_{food} &= F_f \times U \times C_f \\ &= (0.62 \text{ kg/kg-bw/day} \times 1.0 \times 118 \text{ mg/kg}) \\ &= 73.16 \text{ mg/kg-bw/day} \end{aligned}$$

where:

- F_f = Total daily feeding rate in kg food/kg bw of ROC/day (wet basis)
(Table 7-4)
- U = Habitat usage factor (fraction of habitat range represented by site) for
receptor; assumed to be 1.0 for this food web
- C_f = Concentration of COPC in surface soil

The dose from incidental soil is calculated using:

$$\begin{aligned} Dose_{soil} &= F_s \times U \times C_s \\ &= (.062 \text{ kg/kg-bw-day} \times 1.0 \times 118 \text{ mg/kg}) \\ &= 7.32 \text{ mg/kg-bw/day} \end{aligned}$$

where:

- F_s = Total daily incidental soil feeding rate in kg soil/kg-bw of ROC/day (wet
basis) (Table 7-4)
- U = Habitat usage factor (fraction of habitat range represented by site) for
receptor; assumed to be 1.0 for this food web
- C_s = Concentration of COPC in soil; mg chemical/kg soil (dry basis)

The final dose is calculated as follows:

$$\begin{aligned} Dose_{total} &= Dose_{soil} + Dose_{food} \\ Dose_{total} &= 7.32 + 73.16 \\ Dose_{total} &= 80.48 \text{ mg/kg-bw/day (Appendix J, Table J-3)} \end{aligned}$$

The hazard quotient is calculated from the dose and the NOAEL-based TRV as follows:

$$HQ = \frac{DOSE}{TRV_{NOAEL}}$$

$$HQ = \frac{80.48 \text{ mg/kg/day}}{17.58 \text{ mg/kg/day}}$$

$$HQ = 4.6 \text{ (Table 7-9)}$$

7.6 RISK CHARACTERIZATION

7.6.1 Terrestrial Plants

Risk to terrestrial plants is based on a calculation of an Ecological Quotient:

$$\text{Ecological Quotient} = \text{Maximum Surface Soil Concentration} / \text{Reference Toxicity Value}$$

Reference Toxicity Values for plants are the toxicological benchmarks for terrestrial plants provided by Oak Ridge National Laboratory (ORNL) (Efroymson et al. 1997a). The results of the plant assessment are displayed in Table 7-7. If the Maximum Surface Soil Concentration is less than the Reference Toxicity Value, then the Ecological Quotient will be less than 1.0. In this circumstance, no risk is inferred to the plant. If the Maximum Surface Soil Concentration is greater than the Reference Toxicity Value, then the Ecological Quotient will be greater than 1.0, and this carries the inference of potential risk. Ecological Quotients for aluminum, chromium, lead, vanadium, and zinc exceeded the presumed risk threshold of 1.0. When the small exceedances of lead (EQ = 2.4), vanadium (EQ = 9.1), and zinc (EQ = 3.6) are viewed against the nature of the toxicological data, the risk of these COPC to terrestrial plants appears negligible. Laboratory toxicological experiments often involve use of very soluble forms of metals. As noted in Efroymson et al. (1997a); "More commonly, the concentrations reported are nominal concentrations of a soluble form (i.e., a highly bioavailable form) of the chemical added to soil. Most metals in natural soils and contaminants of waste sites are in poorly available forms." For example, most experimental studies of the effects of lead on plants use the relatively soluble form of $PbCl_2$ (Efroymson et al. 1997a). However, in the natural environment, lead typically occurs as sulfite, oxide, carbonite, or sulfate forms (Bodek et al. 1988), which are much

less soluble—and thus, less bioavailable—relative to lead chloride. Therefore, Reference Toxicity Values based on such studies will over predict risk.

The EQ for aluminum was 188. It is pertinent to note that the Reference Toxicity Value for aluminum (50 mg/kg) was based on one lone study that does not allow a high degree of confidence in the benchmark (Efroymson et al. 1997a). The study was based on a seedling establishment of white clover with a reduction of 30% by the addition of 50 mg/kg of aluminum sulfate.

The EQ for chromium was 10 and the Reference Toxicity Value of 1 mg/kg generates low confidence because of the small number of studies on which it is based ((Efroymson et al. 1997a).

Given the low level of exceedence of the Reference Toxicity Values for lead, vanadium, and zinc, any risk to plants from these metals at PI-1 is considered to be negligible. Chromium and aluminum exposure point concentrations were 10 times and 188 times the benchmarks, respectively, representing the most significant risk. It should be noted that mean and maximum concentrations for aluminum and chromium used in this calculation were below the respective mean and 95 % UCLM of the surface soil background values.

7.6.2 Soil Invertebrates

Risk to soil invertebrates was determined using the earthworm as a representative species for the guild. Risk to the earthworm is based on calculation of an Ecological Quotient:

$$\text{Ecological Quotient} = \text{Maximum Surface Soil Concentration} / \text{Reference Toxicity Value}$$

The Reference Toxicity Values are from Efroymson et al. (1997b). If the Maximum Surface Soil Concentration is less than the Reference Toxicity Value, then the Ecological Quotient will be less than 1.0. In this circumstance, no risk is inferred to the earthworm. If the Maximum Surface Soil Concentration is greater than the Reference Toxicity Value, then the Ecological Quotient will be greater than 1.0, and this carries the inference of potential risk.

The results of the earthworm assessment at Site PI-1 are shown in Table 7-8. In the case of all COPC with the exception of chromium, all Maximum Surface Soil Concentrations are lower

than the Reference Toxicity Values. Thus, the Ecological Quotients are all less than 1.0. Chromium has an EQ equal to 25. The benchmark of chromium was established because the 10th percentile lies between the EC50 values of 12 and 15 mg/kg. Confidence in this benchmark is high because of the relatively large amount of data available for a variety of functional measures (Efroymson et al. 1997b). Consequently, potential risk to the earthworm (soil invertebrates) are projected from chromium at PI-1. It should be noted the mean and maximum chromium concentrations at PI-1 are below the respective mean or 95 % UCLM of the surface soil background value for chromium.

7.6.3 Food-Web Risk

The following food web risk characterizations include several assumptions. It is assumed that prey items exhibit the same dry-weight concentration as the maximum soil concentration of COPCs on the site. In addition, wet weight consumption quantities were used with dry-weight soil concentrations. COPC are assumed to be 100 percent bioavailable. That is all of the COPC are available for absorption and expression of toxic effects. Another assumption specific to the food-web analysis is that the area use factor is 1.0. This means that 100 percent of the dietary exposure would be from PI-1. This last assumption is conservative for ROC with large habitat ranges such as an American kestrel, but is more realistic for ROC with small habitat ranges such as the shrew. These assumptions are conservative and contribute to the conservative nature of the risk characterization and to probable overestimation of risk. Hazard quotients for terrestrial receptors are summarized in Table 7-9.

7.6.3.1 Avian Species

The potential impact to avian receptors from exposure to bioconcentratable COPCs via the food web was another assessment endpoint. For this assessment, the ROC were defined as the American robin and American kestrel. In the case of COPC, terminal predators are potentially at risk from the concentrations in the soil in and around PI-1. Species at these higher trophic levels are potentially exposed to toxic substances through the food web as the chemicals proceed upward via magnification. The potential hazards were characterized through comparisons of exposure concentrations (defined as the maximum concentrations of the COPCs at PI-1 converted to dietary doses) to the NOAEL TRVs, listed in Table 7-5.

The risk from exposure of the American robin to COPCs is presented in Appendix J, Table J-1. Six inorganic COPCs were determined to pose potential risks to the robin through dietary exposure. Aluminum was calculated to be 185 times the NOAEL TRV. Chromium exposure was calculated to be 22 times the NOAEL TRV. The HQ of copper was 1.5, lead was 66, vanadium was 3.5, and zinc was 27. It should be noted that the mean and maximum concentrations of aluminum, chromium, and vanadium were below their respective means and 95% UCLM of the surface soil background concentrations. Therefore, the risk of the COPCs aluminum, chromium, and vanadium to the robin is no greater than if the robin foraged in areas outside of PI-1 but in the vicinity. The COPCs lead and zinc, with HQs of 66 and 27, respectively, may pose potential risk to the American robin through dietary exposure.

The risk from exposure of the American kestrel to COPCs is presented in Appendix J, Table J-2. Four inorganic COPCs were determined to pose potential risks to the kestrel through dietary exposure. Aluminum exposure was calculated to be 27 times the NOAEL, chromium was 3.2 times, lead was 9.7 times, and zinc was 3.9 times. Again, the aluminum and chromium mean and maximum concentrations used in the dietary calculations are below the respective mean and 95% UCLM of the surface soil background. The COPCs, lead and zinc, may pose low risk to the American kestrel through dietary exposure.

7.6.3.2 Mammalian Species

The potential impact to mammalian species from exposure to bioconcentratable COPCs via the food web was assessed. For this assessment, the ROC were defined as the short-tailed shrew, meadow vole, and Eastern cottontail. These representative species are potentially at risk from the concentrations in the soil in and around PI-1. Species at these higher trophic levels are potentially exposed to toxic substances through the food web as the chemicals proceed upward via magnification. The potential hazards were characterized through comparisons of exposure doses to the NOAEL TRVs, listed in Table 7-5.

The risk from exposure of the short-tailed shrew to COPCs is presented in Appendix J, Table J-3. Four inorganic COPCs and Total PAH were determined to pose potential risks to the shrew through dietary exposure. Aluminum exposure was calculated to be 2796 times the NOAEL, antimony was 3.8 times, lead was 4.9 times, vanadium was 29 times, and total PAH was 1.1 times. Total PAH was defined as the summation of the PAH COPC and screened against the NOAEL TRV value for benzo(a)pyrene. Site aluminum and vanadium mean and maximum

concentrations were below the respective mean and 95% UCLM for background. The COPCs, antimony, lead, and total PAH have low HQ values and may pose minimal risk to the short-tailed shrew due to dietary exposure.

The risk from exposure of the meadow vole to COPCs is presented in Appendix J, Table J-4. Four inorganic COPCs were determined to pose potential risks to the vole thorough dietary exposure. Aluminum exposure was high, 1923 times the NOAEL values, vanadium was 19.95 times, while antimony and lead were low, 2.59 and 3.15, respectively. Antimony and lead have low HQ values and may pose low risk to the meadow vole through dietary exposure.

The risk from exposure of the Eastern cottontail to COPCs is presented in Appendix J, Table J-5. Four inorganic COPCs were determined to pose potential risks to the rabbit thorough dietary exposure. Aluminum exposure was calculated to be 1164 times the NOAEL value and vanadium was 12 times. Both vanadium and aluminum mean and maximum concentrations used in the risk calculation are below the respective mean and 95% UCLM for background. Antimony exposure was calculated to be 1.6 times the NOAEL and lead was 1.9 times the NOAEL. Antimony and lead have low HQ values and may pose low risk to the Eastern cottontail through dietary exposure.

The risk from exposure of the red fox to COPCs is presented in Appendix J, Table J-6. Four inorganic COPCs were determined to pose potential risks to the rabbit thorough dietary exposure. Aluminum exposure was calculated to be 2458 times the NOAEL value and vanadium was 25 times. Both vanadium and aluminum concentrations used in the risk calculation are below the respective mean and 95% UCLM for background. Antimony exposure was calculated to be 3.3 times the NOAEL and lead was 4 times the NOAEL. Antimony and lead have low HQ values and may pose low risk to the red fox through dietary exposure.

7.7 UNCERTAINTY

Ecological risk characterization includes analysis of uncertainty (EPA 1997c). Uncertainty is distinguished from variability, and arises from lack of knowledge about factors associated with the study. In a screening-level assessment such as this one, uncertainty typically stems from two study facets: the sampling plan and the toxicological data. Sources of uncertainty can include the process of selecting COPC, assumptions made in establishing the Conceptual Site Model, adequacy of ecological characterization of the site, estimates of toxicity to receptors, and

selection of model parameters. There are a number of factors that contribute to uncertainty in the ecological risk characterization for PI-1, as described below.

- Environmental media at known or suspected waste sites are typically sampled in a non-random fashion. That is, sampling points are chosen to best characterize known or suspected areas of contamination. Peripheral and nearby areas are undersampled, if at all, and thus the average exposure of ecological receptors is biased high.
- Toxicological data used in the risk characterization represents significant uncertainty. Because there are no known data on the effects of chemical constituents on the ROC, toxicological data for surrogate species are used, and this adds uncertainty.
- The assumption that total food-web exposure from PI-1 (Area Use Factor=1.0) represent significant uncertainty particularly for receptors with large habitat ranges.
- Food item concentrations were overestimated. Plant and animal food items had not been sampled at the site, and no bioaccumulation factors were used to estimate the chemical concentrations in food items. With the exception of mercury, none of the identified COPC are bioaccumulative. The extremely conservative assumption was made that all food (vegetation, soil invertebrates, mice, etc.) was at the same concentration as the dry weight soil maximum. Based on a review of published bioaccumulation factors for many of the COPC identified in this assessment, actual concentrations in food items are at least a tenth of soil concentration, and often in the hundredths of soil concentrations.
- Food item concentrations were expressed on a dry-weight basis. The food ingestion rates used from the Wildlife Exposure Factors Handbook (EPA 1993b) are ingested food on a wet weight basis. Because dry weight basis soil was directly applied as food concentrations for food items, the exposure to receptors which consume plants and animals (vole, shrew, robin, rabbit, and kestrel) was overestimated. Percent moisture in food items is commonly 50 percent or greater; thus the use of dry-weight food results in an artificial increase of chemical ingestion of at least 100 percent.
- Incidentally ingested soil concentrations are expressed on a dry-weight basis. EPA (1993b) clearly notes that the fraction of incidental soil ingestion should be on a wet



weight basis, and recommends that the wet food ingestion rate be converted to a dry food ingestion rate prior to calculation of dose. This was not performed for this assessment, thus overestimating incidental soil ingestion.

- COPC were assumed to be 100 percent bioavailable. The assumption that COPC are 100 percent bioavailable is highly unlikely based on soil chemistry. Elements such as lead, manganese, and zinc are common constituents of soil and crustal materials. In the solid soil matrix, most of these elements are not bioavailable, and are thus not taken up into organisms exposed to these soils. The environmental behavior (and thus the bioavailability) of metals in environmental soils is complex and not well understood. The solubility and availability of these metals is dependent on a number of factors including soil Eh, pH, and availability of ligands (chemical constituents capable of bonding with metal ions) (Bodek et al. 1988).
- The toxicological data that underpin the screening values is inherently uncertain because laboratory data are extrapolated to specific field sites such as PI-1. This uncertainty is to some extent controlled by choosing the lowest available screening values, consistent with EPA (1997c) guidance to “be consistently conservative in selecting literature values...” This also contributes to overestimation of risk.
- An additional source of uncertainty in this study is related to the variable nature of the swale represented by soil sample SS-9. This sample was collected and designated as soil, although the substrate in this area occasionally is wet. However, regardless of whether this particular sample location was wet or dry, the maximum concentration of each analyte from the sample was screened against soil screening values. Had the station been screened against sediment screening values, the results may have been different since the screening values are different. For example, many sediment screening values for PAHs are higher than soil screening values. If the soil screening values for PAHs were replaced by sediment screening values some COPCs may disappear and most screening indices would be much lower.

Although the direction of bias of some uncertainties is unknown, the influence of the non-random media sampling and assumptions of 100 percent bioavailability and site exposure assures that, if anything, risk is overestimated rather than underestimated.

7.8 SUMMARY OF ECOLOGICAL RISK SCREENING

The ecological risk screening process identified 21 COPCs (Table 7-3). By definition, any Screening Quotient greater than 1.0 represents potential risk from that analyte in a screening-level assessment. The identification of these ecological COPCs represents the culmination of Steps 1 and 2 of EPA's ecological risk assessment guidance (EPA 1997c). A summary of risk due to dietary exposure for the ROC is presented in Table 7-9. Several COPCs had HQ values above one including aluminum, antimony, chromium, copper, lead, vanadium, zinc, and Total PAH. It should be noted that the mean and maximum concentrations used in the food web modeling for aluminum, chromium, and vanadium are below the respective mean and 95% UCLM of the surface soil background values reported for Fort Pickett (WESTON 1999a). This indicates the ROC could ingest soil at PI-1 and the risk would be below the risk from ingestion of background surface soil samples. The COPCs antimony, copper, lead, zinc, and Total PAH pose a potential risk to several of the ROC. A refined food web is recommended for the seven metals (aluminum, antimony, chromium, copper, lead, vanadium, zinc) and Total PAH that resulted in hazards greater than 1.0 in the simplified risk assessment. Factors to be considered in this refined food web include:

- Use of proper Area Use Factors for each ROC,
- Use of appropriate exposure concentrations (mean for large habitat ROC, maximum or 95%UCLM for smaller habitat ROC),
- Use of appropriate bioaccumulation factors for these metals and chemicals into ROC,
- Use of proper wet to dry weight conversions for the food web,
- Use of more realistic ROC body weight and ingestion rates,
- Approximation of ROC feeding fractions into the refined food web.

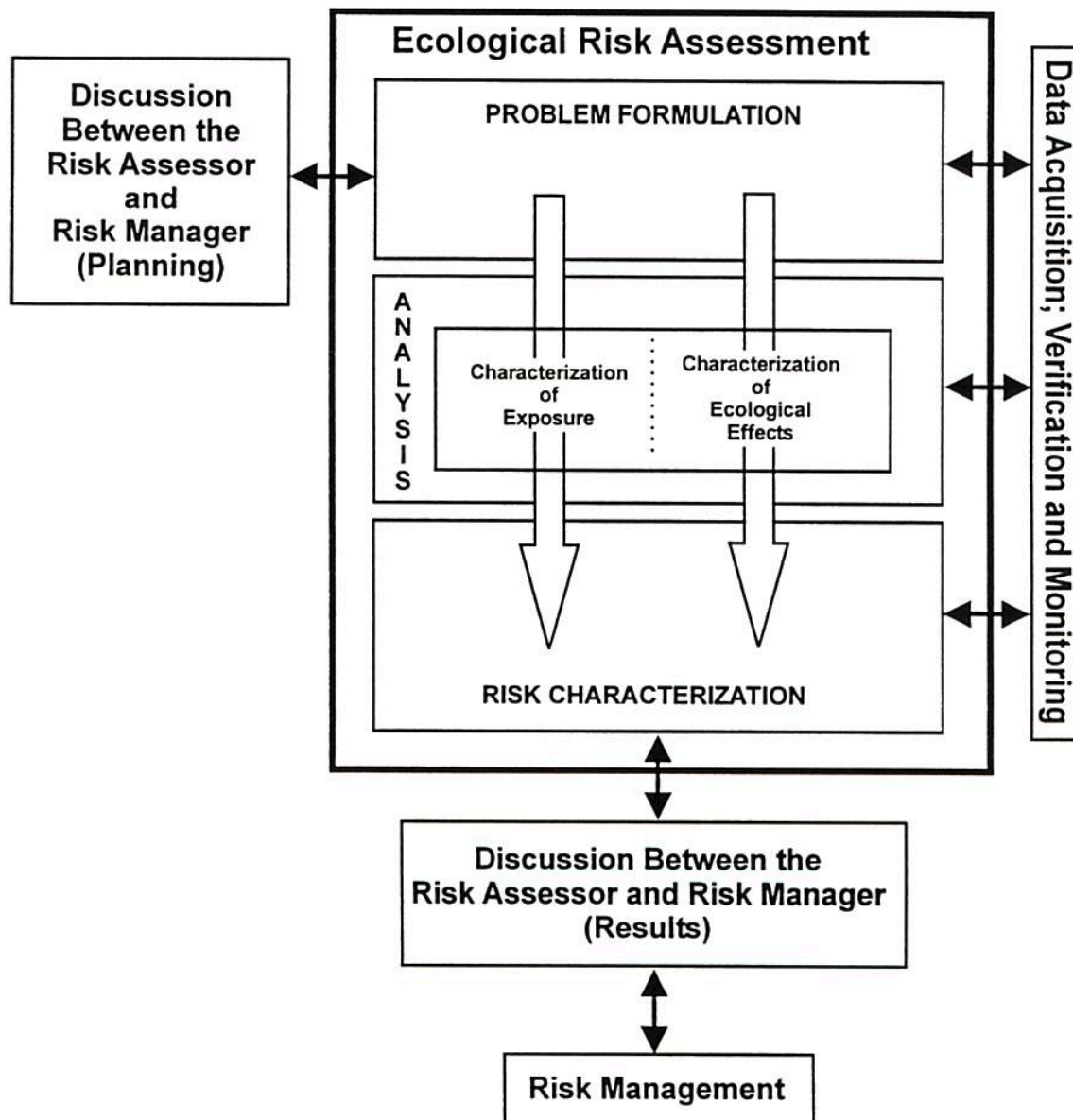


Figure 7-1. The EPA Ecological Risk Assessment Framework.

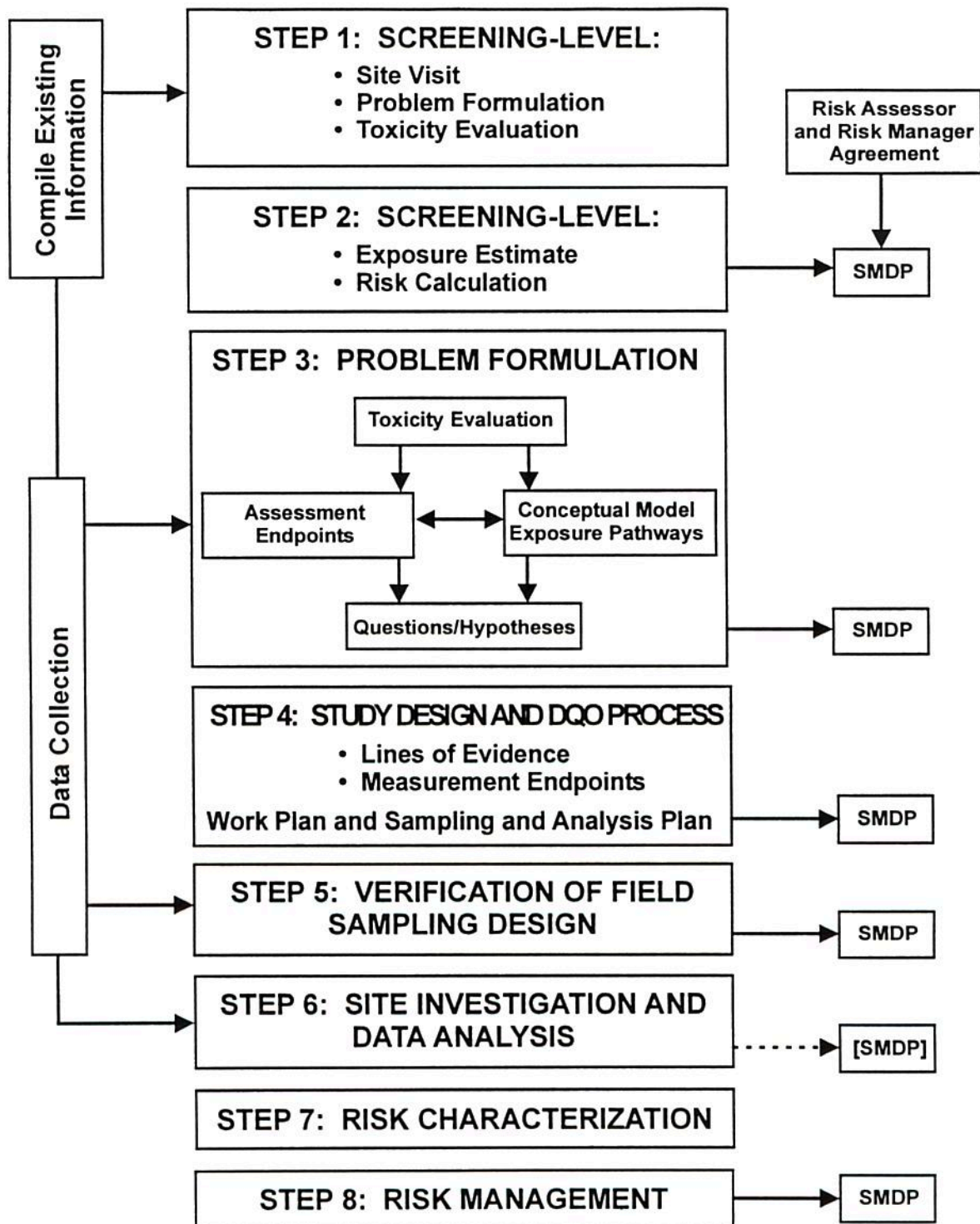


Figure 7-2. The EPA Eight-Step Ecological Risk Assessment Process for Superfund.

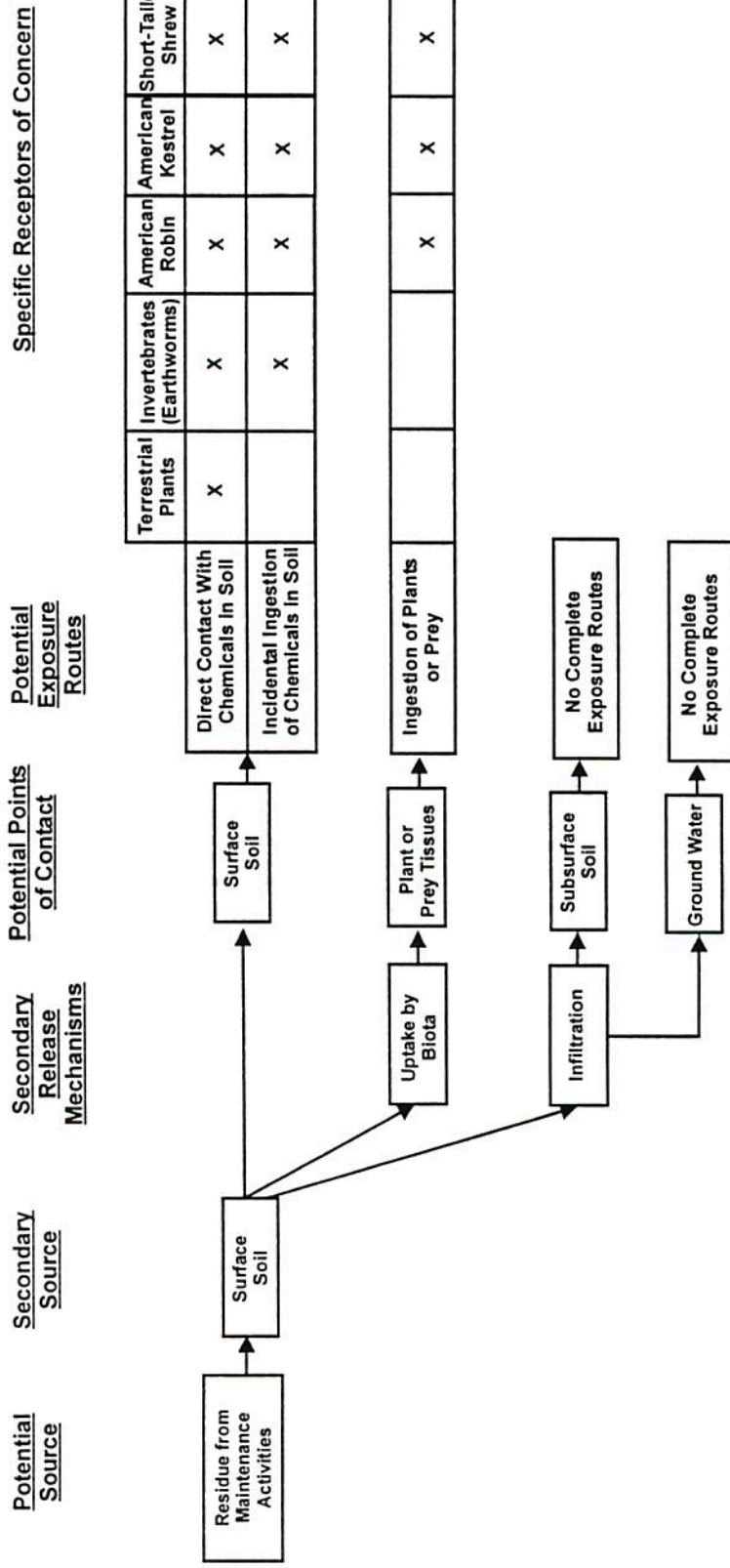


Figure 7-3. Ecological Risk Conceptual Site Model for PI-1, Fort Pickett, Virginia.

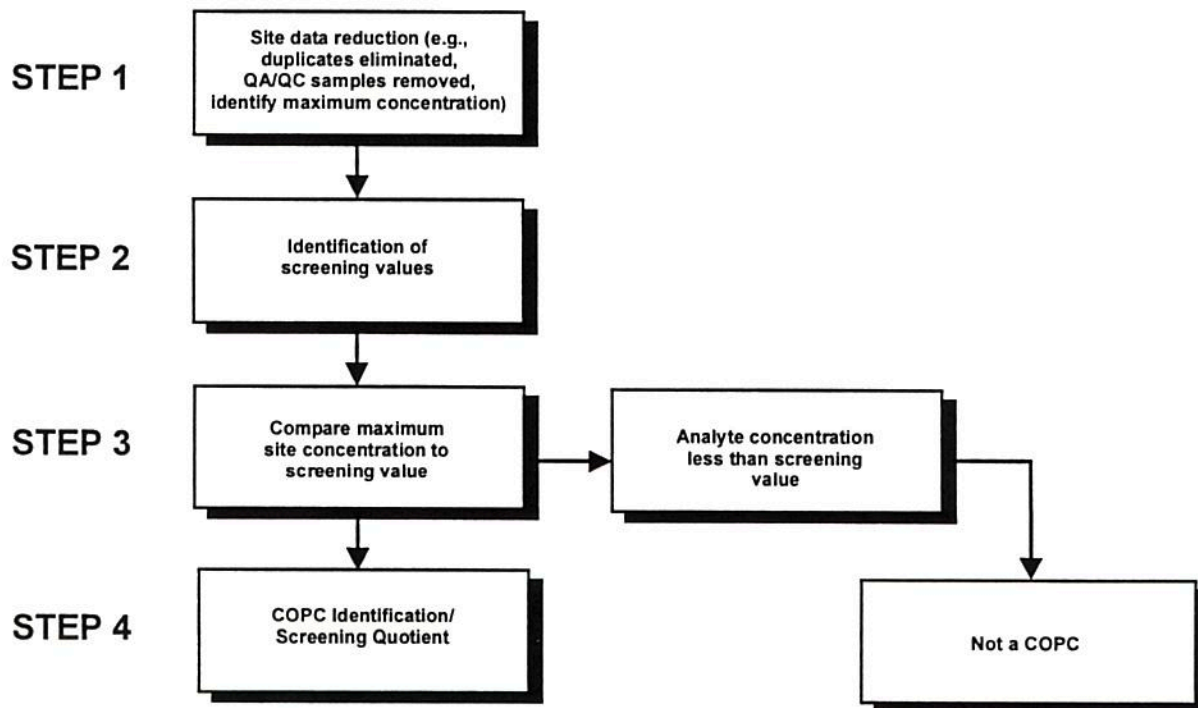


Figure 7-4. Ecological COPC Screening Process.

TABLE 7-1 ECOLOGICAL INVENTORY FOR SITE PI-1

Common Name	Species Name
Vegetation	
Sweet gum	<i>Liquidambar styraciflua</i>
Loblolly pine	<i>Pinus taeda</i>
Yucca	<i>Yucca filamentosa</i>
Alder	<i>Alnus sp.</i>
Greenbrier	<i>Smilax rotundifolia</i>
Christmas fern	<i>Polystichum acrostichoides</i>
Trout lily	<i>Erythronium americanum</i>
Southern Red oak	<i>Quercus falcata</i>
Black oak	<i>Quercus velutina</i>
Turkey-foot oak	<i>Quercus laevis</i>
White oak	<i>Quercus alba</i>
Eastern Red cedar	<i>Juniperus virginiana</i>
American Beech	<i>Fagus grandifolia</i>
Sweetbay magnolia	<i>Magnolia virginiana</i>
Red Maple	<i>Acer rubrum</i>
Birds	
Blue jays	<i>Cyanocitta cristata</i>
Killdeer	<i>Charadrius vociferus</i>
Slate-colored junco	<i>Junco hyemalis</i>
American crow	<i>Corvus brachyrhynchos</i>
Carolina chickadee *	<i>Parus carolinensis</i>
Turkey vulture *	<i>Cathartes aura</i>
Carolina wren *	<i>Thryothorus ludovicianus</i>
Pileated woodpecker *	<i>Dryocopus pileatus</i>
Yellow-bellied sapsucker *	<i>Sphyrapicus varius</i>
American Robin *	<i>Turdus migratorius</i>
Northern cardinal *	<i>Cardinalis cardinalis</i>
Mourning dove *	<i>Zenaida macroura</i>
Common Flicker *	<i>Colaptes auratus</i>
Downey woodpecker *	<i>Picoides pubescens</i>

TABLE 7-1 (Continued)

Common Name	Species Name
Tufted Titmouse *	<i>Parus bicolor</i>
House sparrow *	<i>Passer domesticus</i>
European Starling *	<i>Sturnus vulgaris</i>
American goldfinch *	<i>Carduelis tristis</i>
White-breasted Nuthatch *	<i>Sitta canadensis</i>
Northern Mockingbird *	<i>Mimus polyglottos</i>
American Kestrel *	<i>Falco sparverius</i>
Cooper's hawk *	<i>Accipiter gentilis</i>
Hairy woodpecker *	<i>Picoides villosus</i>
Red-bellied woodpecker *	<i>Melanerpes carolinus</i>
Chipping sparrow *	<i>Spizella passerina</i>
Song sparrow *	<i>Melospiza melodia</i>
Reptiles and Amphibians	
American Toad *	<i>Bufo americanus</i>
Spring peeper *	<i>Hyla crucifer</i>
Slimy salamanders *	<i>Plethodon glutinosus</i>
Eastern newt *	<i>Notophthalmus viridescens</i>
Box Turtles *	<i>Terrapene carolina</i>
Garter snake *	<i>Thamnophis viridescens</i>
Mammals	
White-tailed deer	<i>Odocoileus virginianus</i>
Eastern gray squirrel *	<i>Sciurus carolinensis</i>
Meadow voles *	<i>Microtus pennsylvanicus</i>
Short-tailed shrew *	<i>Blarina brevicauda</i>
Striped skunk *	<i>Mephitis mephitis</i>
Eastern cottontail *	<i>Sylvilagus floridanus</i>

* = presumed present

TABLE 7-2. ECOLOGICAL RISK SCREENING ASSESSMENT ENDPOINTS AT PI-1

Assessment Endpoint	Null Hypothesis	Measurement Endpoint	Specifics of Assessment
Ecological health of terrestrial plant communities	Soils are not exhibiting a detrimental effect on plant survival and growth	Evaluation of soil chemistry with respect to vegetation screening values	<ul style="list-style-type: none"> Comparison of soil concentrations to vegetation screening values
Ecological health of terrestrial invertebrate communities	Soils are not exhibiting a detrimental effect on invertebrate survival and growth	Evaluation of soil chemistry with respect to soil invertebrate screening values	<ul style="list-style-type: none"> Comparison of soil concentrations to soil invertebrate screening values
Long term health and reproductive capacity of omnivorous avian species (American robin)	Ingestion of COPC in prey does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soils data and dietary exposure models	<ul style="list-style-type: none"> Vegetation and invertebrate dose approximated by assuming equivalence to soil concentration The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).
Long term health and reproductive capacity of carnivorous avian species (American kestrel)	Ingestion of COPC in prey does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soils data and dietary exposure models	<ul style="list-style-type: none"> Small mammal dose approximated by assuming equivalence to soil concentration The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).
Long term health and reproductive capacity of small herbivorous mammalian species (Meadow vole)	Ingestion of COPC in food does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soils data and dietary exposure models	<ul style="list-style-type: none"> Vegetation dose approximated by assuming equivalence to soil concentration The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).

TABLE 7-2 (Continued)

Assessment Endpoint	Null Hypothesis	Measurement Endpoint	Specifics of Assessment
Long term health and reproductive capacity of small carnivorous mammalian species (Short-tailed shrew)	Ingestion of COPC in food does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soils data and dietary exposure models	<ul style="list-style-type: none"> • Soil invertebrate dose approximated by assuming equivalence to soil concentration • The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).
Long term health and reproductive capacity of medium-sized herbivorous mammalian species (Eastern cottontail)	Ingestion of COPC in prey does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soil data and dietary exposure models	<ul style="list-style-type: none"> • Dose from vegetation approximated by assuming equivalence to soil concentration • The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).
Long term health and reproductive capacity of large-sized carnivorous mammalian species (red fox)	Ingestion of COPC in prey does not have a negative impact on growth, survival, and reproductive success of the species	Evaluation of dose in prey based on surface soil data and dietary exposure models	<ul style="list-style-type: none"> • Dose from prey approximated by assuming equivalence to soil concentration • The risk associated with the calculated dose will be evaluated by comparison to Toxicity Reference Values (TRVs).

TABLE 7-3 OCCURRENCE, DISTRIBUTION, AND SELECTION OF ECOLOGICAL COPC IN SURFACE SOIL AT PI-1

Chemical	Minimum Concentration	Qualifier	Maximum Concentration	Qualifier	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Screening Value	Screening Quotient Ratio	COPC	Additional Considerations
DIOXINS												
TEQ (ND=1/2)	8.28		95.95		ng/kg	PII-SB6	4/4		10000	0.01	No	
INORGANICS												
ALUMINUM	1840	*	9410	J	mg/kg	PII-SB4	9/9	1.9 - 4.9	1	9410.00	Yes	
ANTIMONY	0.27	L	0.825	L	mg/kg	PII-SB6	8/9	0.22 - 0.28	0.48	1.72	Yes	
ARSENIC	0.89	B	3.1	L	mg/kg	PII-SB4	9/9	0.09 - 0.12	10	0.31	No	
BARIUM	20.2	B	76.9	J	mg/kg	PII-SB6	9/9	0.91 - 1.2	440	0.17	No	
BERYLLIUM	0.17	B	0.3	J/B	mg/kg	PII-SB3/PII-SB9	9/9	0.01 - 0.01	0.02	15.00	Yes	
CADMIUM	0.25	B	1		mg/kg	PII-SB8/PII-SB9	8/9	0.02 - 0.02	2.5	0.40	No	
CALCIUM	684	B	127000		mg/kg	PII-SB8	9/9	6.1 - 7.9	N/A		No	Essential
CHROMIUM	4.6		10		mg/kg	PII-SB9	9/9	0.09 - 0.4	0.0075	1333.33	Yes	
COBALT	0.85	L	2	L	mg/kg	PII-SB6	4/9	0.27 - 0.85	20	0.10	No	
COPPER	1.4	J	33.1		mg/kg	PII-SB8	9/9	0.08 - 0.2	15	2.21	Yes	
IRON	4910		14200	J	mg/kg	PII-SB4	9/9	0.86 - 17.2	N/A		No	Essential
LEAD	11.5		118		mg/kg	PII-SB4/PII-SB9	9/9	0.09 - 0.12	0.01	11800.00	Yes	
MAGNESIUM	253	J	252600		mg/kg	PII-SB6	9/9	3.6 - 11.5	N/A		No	Essential
MANGANESE	42.3		131	J	mg/kg	PII-SB3	9/9	0.28 - 0.89	100	1.31	Yes	
MERCURY	0.06	B	0.1	B	mg/kg	PII-SB8	3/9	0.04 - 0.07	0.058	1.72	Yes	
NICKEL	1.1	B	9.1		mg/kg	PII-SB8	7/9	0.26 - 1.5	2	4.55	Yes	
POTASSIUM	232		1450		mg/kg	PII-SB5	9/9	23.2 - 29.2	N/A		No	Essential
THALLIUM	0.14	B	0.23	B	mg/kg	PII-SB9	5/9	0.11 - 0.14	0.001	230.00	Yes	
VANADIUM	5.4	*	18.2	*	mg/kg	PII-SB4	9/9	0.74 - 0.93	0.5	36.40	Yes	
ZINC	4.1	J	181		mg/kg	PII-SB8	9/9	0.26 - 1.1	10	18.10	Yes	
PAHs												
2-METHYLNAPHTHALENE	5.6		660		ug/kg	PII-SB7	9/9	1.8 - 86	100	6.60	Yes	
ACENAPHTHENE	1.9		12	J	ug/kg	PII-SB7	6/9	1.8 - 2.2	100	0.12	No	
ACENAPHTHYLENE	2.1	J	14	J	ug/kg	PII-SB7	5/9	1.8 - 2.2	100	0.14	No	
ANTHRACENE	2.4	J	19	J	ug/kg	PII-SB7	6/9	1.8 - 2.2	100	0.19	No	
BENZ[A]ANTHRACENE	2		87	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	0.87	No	
BENZO[G,H]PERYLENE	2.4		77	J	ug/kg	PII-SB7	8/9	1.8 - 2.2	100	0.77	No	
BENZO[A]PYRENE	1.9		85	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	0.85	No	
BENZO[B]FLUORANTHENE	4.4		150	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	1.50	Yes	
BENZO[K]FLUORANTHENE	2.5		41	J	ug/kg	PII-SB7	7/9	1.8 - 2.2	100	0.41	No	
CHRYSENE	4.7	J	130	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	1.30	Yes	
DIBENZ[A,H]ANTHRACENE	1.85	UJ	20	J	ug/kg	PII-SB7	4/9	1.8 - 2.2	100	0.20	No	
FLUORANTHENE	2.8		110	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	1.10	Yes	
FLUORENE	2.9	J	26	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	0.26	No	
INDENO[1,2,3-C,D]PYRENE	2.3		62	J	ug/kg	PII-SB7	8/9	1.8 - 2.2	100	0.62	No	
NAPHTHALENE	4.5		270	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	2.70	Yes	
PHENANTHRENE	8.8		360	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	3.60	Yes	
PYRENE	4.1	J	270	J	ug/kg	PII-SB7	9/9	1.8 - 2.2	100	2.70	Yes	

TABLE 7-3 OCCURRENCE, DISTRIBUTION, AND SELECTION OF ECOLOGICAL COPC IN SURFACE SOIL AT PI-1

Chemical	Minimum Concentration	Qualifier	Maximum Concentration	Qualifier	Units	Location of Maximum Concentration	Detection Frequency	Range of Detection Limits	Screening Value	Screening Quotient Ratio	COPC	Additional Considerations
PESTICIDES/PCB												
ALPHA-CHLORDANE	3.4		3.4		ug/kg	PII-SB7	1/9	0.74 - 0.95	100	0.03	No	
DDD	0.79	J	8.7		ug/kg	PII-SB7	4/9	0.45 - 0.57	100	0.09	No	
DDE	1.4	J	31		ug/kg	PII-SB7	5/9	0.42 - 8.8	100	0.31	No	
DDT	1.5	J	40		ug/kg	PII-SB7	3/9	0.7 - 14	100	0.40	No	
DELTA-HCH	0.89	J	0.89		ug/kg	PII-SB8	1/9	0.52 - 0.66	1000	< 0.01	No	
DIELDRIN	0.79	J	1.3		ug/kg	PII-SB7	3/9	0.46 - 0.58	100	0.01	No	
ENDRIN	2.7	J	2.7		ug/kg	PII-SB8	1/9	1.6 - 2	100	0.03	No	
GAMMA-CHLORDANE	2.7	J	2.7		ug/kg	PII-SB7	1/9	0.37 - 0.47	100	0.03	No	
HEPTACHLOR EPOXIDE	1.1	J	1.7		ug/kg	PII-SB7	2/9	0.86 - 1.1	100	0.02	No	
ENDOSULFAN I	1.8	J	1.8		ug/kg	PII-SB4	1/9	0.76 - 0.98	25000	< 0.01	No	
ENDRIN KETONE	1.3		1.3		ug/kg	PII-SB9	1/9	0.74 - 0.95	30	0.04	No	
SEMIVOLATILES												
BIS(2-ETHYLHEXYL)PHTHALATE	53	J	100		ug/kg	PII-SB5	4/9	56 - 72	30000(a)	< 0.01	No	
BUTYLBENZYLPHTHALATE	120		120		ug/kg	PII-SB8	1/9	59 - 76	30000(a)	< 0.01	No	
DIBENZOFURAN	180		180		ug/kg	PII-SB7	1/9	77 - 99	N/A		Yes	NSV
VOLATILES												
ACETONE	21		750		ug/kg	PII-SB5	9/9	2 - 3	N/A		Yes	NSV
M&P XYLENES	5	L	5		ug/kg	PII-SB8	1/9	1 - 1	100	0.05	No	
O-XYLENE	4	L	4		ug/kg	PII-SB8	1/9	0.9 - 1	100	0.04	No	

N/A = Not Applicable
COPC = Chemical of Potential Concern
NSV = No Screening Values are available
* = Duplicate analysis is not within control limits
B = Reported Value is less than the CRDL but greater than the IDL (inorganic)
J = Indicates an estimated value
L = Biased low
U = Sample recovery not within control limits (inorganic)
U = Analyte analyzed for but not detected

(a) Based on total phthalates

TABLE 7-4 EXPOSURE FACTORS FOR SITE PI-1 TERRESTRIAL ECOLOGICAL RECEPTORS OF CONCERN

Exposure Factor	Receptor				
	American Robin	American Kestrel	Meadow Vole	Short-Tailed Shrew	Eastern Cottontail
Body Weight (kg)	0.0635 (EPA 1993b) Lowest range for adult, both sexes, all seasons, PA location	0.103 (EPA 1993b) Lowest value from CA, UT studies	.0355 (EPA 1993b) Adult male average, all year, IN location	0.0125 (EPA 1993b) Lowest value in range from PA study	0.801 (EPA 1993b) Lowest value in range for adult male, MD, WV populations
Food Ingestion Rate (g/g/day)	1.96 (EPA 1993b) Highest value in the range for both sexes, free-living in KS location	0.31 (EPA 1993b) Highest value for adult male in summer, OH location	0.35 (EPA 1993b) Highest value for adults, Russia	0.62 (EPA 1993b) Highest value for adults, both sexes, WI	0.093 Calculated using equation for herbivores in EPA (1993b)
Incidental Soil Ingestion Rate (g/g/day)	0.20 Based on 10.4% soil in diet of American woodcock, (EPA 1993b)	0.0047 No data available; assumed low rate of 1.5% soil in diet based on food items and capture mode.	0.0084 Based on 2.4% soil in diet of meadow vole, (EPA 1993b)	0.062 No data available; a relatively high rate [10% soil in diet] was assumed because of the preponderance of earthworms and soil invertebrates in diet.	0.00392 Based on 2.8 percent soil in diet of red fox, (EPA 1993b)

TABLE 7-5 LIST OF TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING
(using Estimated Wildlife NOAEL [mg/kg-bw/day])

COPC	Shrew	Robin	Rabbit	Kestrel	Vole	Red Fox	Comments
INORGANICS							
Aluminum	2.295	109.7	0.767	109.7	1.754	0.551	NOAEL for vole, shrew, fox, and rabbit based on mouse exposed to AlCl ₃ . NOAEL for robin and kestrel based on dove exposed to aluminum sulfate.
Antimony	0.149	NA	0.05	NA	0.114	0.036	NOAEL for vole, shrew, fox, and rabbit based on mouse exposed to antimony potassium tartrate.
Beryllium	1.45	NA	0.49	NA	1.11	0.35	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL for beryllium sulfate.
Chromium	7.21	1	2.41	1	5.51	1.73	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL for Cr+6. NOAEL for robin and kestrel based on duck NOAEL for Cr+3.
Copper	33.4	47	11.2	47	25.5	8	NOAEL for vole, shrew, fox, and rabbit based on mink NOAEL. NOAEL for robin and kestrel based on chick NOAEL.
Lead	17.58	3.85	5.88	3.85	13.44	4.22	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on kestrel NOAEL.
Manganese	193	997	65	997	148	46	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on quail NOAEL.
Mercury	2.86	0.45	0.96	0.45	2.18	0.69	NOAEL for shrew, vole, rabbit, and fox based on mink mercuric chloride NOAEL. NOAEL for robin and kestrel based on quail mercuric chloride NOAEL.

TABLE 7-5 LIST OF TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING
(using Estimated Wildlife NOAEL [mg/kg-bw/day])

COPC	Shrew	Robin	Rabbit	Kestrel	Vole	Red Fox	Comments
Nickel	87.91	77.4	29.4	77.4	67.18	21.12	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on mallard NOAEL.
Thallium	0.016	NA	0.005	NA	0.013	0.004	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL.
Vanadium	0.428	11.4	0.143	11.4	0.327	0.103	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on mallard NOAEL.
Zinc	351.7	14.5	117.6	14.5	268.7	84.5	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on chicken NOAEL.
PAH							
Total PAH	1.19	NA	0.4	NA	0.91	0.29	NOAEL value used is for benzo(a)pyrene based on mouse NOAEL.
SEMIVOLATILES							
Dibenzofuran	NA	NA	NA	NA	NA	NA	
VOLATILES							
Acetone	22	NA	7.3	NA	16.8	5.3	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL.

NA = Not Available

NOAEL values for the Red-tailed hawk used for the American kestrel, red fox NOAEL values used for raccoon (Sample et al. 1996)

TABLE 7-6 LIST OF TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING
(using Estimated Wildlife LOAEL [mg/kg-bw/day])

COPC	Shrew	Robin	Rabbit	Kestrel	Vole	Red Fox	Comments
INORGANICS							
Aluminum	22.952	1097	7.674	1097	17.538	5.515	LOAEL for vole, shrew, fox, and rabbit based on mouse LOAEL exposed to AlCl ₃ . LOAEL for robin and kestrel equals NOAEL x 10.
Antimony	1.487	NA	0.497	NA	1.136	0.357	LOAEL for vole, shrew, fox, and rabbit based on mouse LOAEL exposed to antimony potassium tartrate.
Beryllium	14.5	NA	4.9	NA	11.1	3.5	LOAEL equals NOAEL times 10
Chromium	28.88	5	9.66	5	22.07	6.94	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL for Cr+6. LOAEL for robin and kestrel based on duck LOAEL for Cr+3.
Copper	44	61.7	14.7	61.7	33.6	10.6	LOAEL for vole, shrew, fox, and rabbit based on mink LOAEL. LOAEL for robin and kestrel based on chick LOAEL.
Lead	174.83	38.5	58.79	38.5	134.35	42.25	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on kestrel NOAEL times 10.
Mercury	28.6	0.9	9.6	0.9	21.8	6.9	LOAEL for shrew, vole, rabbit and red fox equals mercuric chloride NOAEL times 10. LOAEL for robin and kestrel based on quail mercuric chloride LOAEL.
Manganese	624	9970	209	9970	477	150	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on quail NOAEL times 10.

TABLE 7-6 LIST OF TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING
(using Estimated Wildlife LOAEL [mg/kg-bw/day])

COPC	Shrew	Robin	Rabbit	Kestrel	Vole	Red Fox	Comments
INORGANICS							
Nickel	175.83	107	58.79	107	134.35	42.25	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on mallard LOAEL.
Thallium	0.164	NA	0.055	NA	0.126	0.039	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL.
Vanadium	4.285	114	1.433	114	3.274	1.03	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on mallard NOAEL times 10.
Zinc	703.3	131	235.2	131	537.4	169	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on chicken LOAEL.
PAH							
Total PAH	11.89	NA	3.98	NA	9.09	2.86	LOAEL value used is for benzo(a)pyrene based on mouse LOAEL.
SEMIVOLATILES							
Dibenzofuran	NA	NA	NA	NA	NA	NA	
VOLATILES							
Acetone	109.9	NA	36.7	NA	84	26.4	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL.

LOAEL TRV data from Sample et al. (1996)
LOAEL values for the Red-tailed hawk used for the American kestrel.
NA = Not Applicable

TABLE 7-7. ECOLOGICAL QUOTIENTS FOR TERRESTRIAL PLANTS FOR COPCs AT SITE PI-1

COPC	Surface Soil Maximum (mg/kg)	Reference Toxicity Value (mg/kg)	Ecological Quotient (EQ _{max})
Aluminum	9410	50	188
Antimony	0.825	5	0.165
Beryllium	0.3	10	0.03
Chromium	10	1	10
Copper	33.1	100	0.331
Lead	118	50	2.36
Manganese	131	500	0.262
Mercury	0.1	0.3	0.333
Nickel	9.1	30	0.303
Thallium	0.23	1	0.23
Vanadium	18.2	2	9.1
Zinc	181	50	3.62
Total PAH *	1.95	20	0.098
Dibenzofuran	0.18	NA	NA
Acetone	0.75	NA	NA

Reference Toxicity Values from Efroymson et al. 1997

* Value for acenaphthene used for Total PAH

TABLE 7-8. ECOLOGICAL QUOTIENTS FOR EARTHWORMS FOR COPCs AT SITE PI-1

COPC	Surface Soil Maximum (mg/kg)	Reference Toxicity Value (mg/kg)	Ecological Quotient (EQ _{max})
Aluminum	9410	NA	NA
Antimony	0.825	NA	NA
Beryllium	0.3	NA	NA
Chromium	10	0.4	25
Copper	33.1	60	0.552
Lead	118	500	0.236
Manganese	131	NA	NA
Mercury	0.1	0.1	1
Nickel	9.1	200	0.046
Thallium	0.23	NA	NA
Vanadium	18.2	NA	NA
Zinc	181	200	0.905
Total PAH *	1.95	30	0.065
Dibenzofuran	0.18	NA	NA
Acetone	0.75	NA	NA

Reference Toxicity Values from Efroymson et al. 1997

* Value for fluorene used for Total PAH

TABLE 7-9. FORT PICKETT, SITE PI-1 HAZARD QUOTIENT SUMMARY

TERRESTRIAL SPECIES
MAXIMUM CONCENTRATION HAZARD QUOTIENT VALUES
FORT PICKETT, SITE PI-1

Ecological Contaminants of Concern	Shrew		Vole		Rabbit		Robin		Kestrel		Red Fox	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
	HQ _n	HQ _i	HQ _n	HQ _i	HQ _n	HQ _i	HQ _n	HQ _i	HQ _n	HQ _i	HQ _n	HQ _i
Aluminum	2796	280	1923	192	1164	116	185	457	27	67	2458	246
Antimony	3.78	0.38	2.59	2.59	1.57	0.16	NA	NA	NA	NA	3.30	0.33
Beryllium	0.14	0.01	0.10	0.01	0.06	0.01	NA	NA	NA	NA	0.12	0.01
Chromium	0.95	0.24	0.65	0.16	0.39	0.10	21.6	4.32	3.15	0.63	0.83	0.21
Copper	0.68	0.51	0.47	0.35	0.28	0.21	1.52	1.16	0.22	0.17	0.60	0.45
Lead	4.58	0.46	3.15	0.31	1.90	0.19	66.2	6.62	9.65	0.96	4.02	0.40
Manganese	0.46	0.14	0.32	0.10	0.19	0.06	0.28	0.03	0.04	0.00	0.41	0.13
Mercury	0.02	0.00	0.02	0.00	0.01	0.00	0.48	0.24	0.07	0.03	0.02	0.00
Nickel	0.07	0.04	0.05	0.02	0.03	0.01	0.25	0.18	0.04	0.03	0.06	0.03
Thallium	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
Vanadium	29.0	2.90	19.9	1.99	12.1	1.21	3.45	0.34	0.50	0.05	25.4	2.54
Zinc	0.35	0.18	0.24	0.12	0.15	0.07	27.0	2.98	3.93	0.43	0.31	0.15
Total PAH	1.12	0.11	0.77	0.08	0.46	0.05	NA	NA	NA	NA	0.97	0.10
Dibenzofuran	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Acetone	0.02	0.00	0.02	0.00	0.01	0.00	NA	NA	NA	NA	0.02	0.00

NA - Not Available
HQ_n - Hazard Quotient based on the NOAEL
HQ_i - Hazard Quotient based on the LOAEL

8. STEP 3 ECOLOGICAL RISK ASSESSMENT

8.1 INTRODUCTION

The Tier I ecological risk screening process identified 21 COPCs at PI-1. An assessment of risk due to dietary exposure using conservative Step 2 exposure assumptions revealed that several COPCs had Hazard Quotient (HQ) values above 1.0 including aluminum, antimony, chromium, copper, lead, vanadium, zinc, and total polycyclic aromatic hydrocarbons (PAH). A refined food web was recommended for these seven metals and Total PAH. The Step 3 refined food web modeling was agreed to during the 13 June 2000 SMDP by EPA Region 3, the Ft. Pickett BRAC office, and the USACE. Background concentrations of metals may also be discussed relative to site metal concentrations during a Step 3 ERA. Factors considered in this refined food web included:

- Proper wet to dry weight conversions for the food web
- Proper Area Use Factors for each receptor of concern (ROC)
- Appropriate exposure concentrations [mean for large habitat ROC, maximum or 95%UCLM for small habitat ROC]
- Appropriate bioaccumulation factors for these COPCs into ROC
- More realistic ROC body weight and ingestion rates
- Approximation of ROC feeding fractions into the refined food web

8.2 FOOD-WEB MODIFICATIONS

The modified food-web accounts for food and incidental soil ingestion to estimate dose to the ecological receptors. Water dose has not been considered because concentrations of chemicals in water are small relative to concentrations found in soil or food, and would add insignificantly to the total dose. Details regarding food-web modifications follow.

8.2.1 Exposure Factors

Exposure factors have been adjusted to reflect average body weights, food ingestion rates, and incidental soil ingestion rates. These rates were obtained from EPA (1993), and are shown in Table 8-1 for terrestrial ecological ROC. It is important to note that the incidental soil ingestion

rate is in units of grams (g) wet soil/g body weight/day in Table 8-1. As reported from the analytical laboratory, soil data are expressed in dry weight concentrations. Dry weight results reported by the laboratory have been converted to wet weight concentrations based on reported percent moisture from those samples using the following equation:

$$[x]_{\text{wet}} = [x]_{\text{dry}} \times f_{\text{dry}}$$

where:

$[x]_{\text{wet}}$ = concentration of chemical x on a wet weight basis

$[x]_{\text{dry}}$ = concentration of chemical x on a dry weight basis (as reported by the laboratory)

f_{dry} = fraction dry for the sample (as reported by the laboratory)

An area use factor (AUF) has been introduced into the exposure assessment. The AUF is the size of PI-1 (0.33 ha) divided by the territory of the receptor shown in Table 8-1. If the AUF was less than 0.5, the exposure concentration was assumed to be the upper 95%UCLM. The only ROC for which this applied was the meadow vole with a standard habitat range of 0.019 ha. This represents approximately 5 percent of the size of PI-1, thus use of the 95%UCLM is appropriate for the meadow vole. All other ROC were exposed to the arithmetic mean concentration of the COPC.

Each ROC consumes different proportions of types of food. The refined food-web accounted for this based on ROC specific dietary fractions, also shown in Table 8-1.

The Tier I, Step 1 and 2 food-web assumed that COPCs were present in food sources at the same concentration as that found in the soil. For the refined Step 3 food-web, bioaccumulation factors (BAFs) have been applied. The BAF represents that portion of chemical that is transferred from the soil matrix to the prey organism. Appropriate BAFs have been obtained from the scientific literature for the three prey items of soil invertebrates, plants, and small mammals, and are shown in Tables 8-2 through 8-4, respectively.

8.2.1.1 Soil Invertebrate Bioaccumulation

Sample et al. (1998a) published guidance for determining bioaccumulation factors for metals in earthworms, here used as a surrogate for soil invertebrates. A general equation was developed to

describe the relationship between earthworm concentrations and soil concentrations (Table 12 in Sample et al. 1998a). These regression equations have been used to predict soil invertebrate concentrations of copper, lead, and zinc. These regression equations predict the dry organism concentration, and these were converted to wet weight organism bioaccumulation assuming that invertebrates are 84 percent water (EPA 1993). The regression equations are as follows:

$$\ln[\text{earthworm}] = B_0 + B_1 (\ln[\text{soil}])$$

where:

B_0 and B_1 are contaminant specific parameters

$\ln[\text{soil}]$ is the natural log of the COPC concentration in soil

$\ln[\text{earthworm}]$ is the natural log of the calculated earthworm concentration

The earthworm concentrations (dry weight) were converted to wet weight by multiplying by the fraction dry of earthworms (assumed to be 16 percent based on 84 percent moisture value). The results of the regression calculations are presented in Table 8-6.

Some metals had insufficient toxicological data to develop these equations. Therefore, the mean uptake factors (UF) from Table 11 or Appendix C in Sample et al. (1998a) were used for aluminum, chromium, and vanadium. Antimony was not investigated by Sample et al. (1998a); therefore, Baes et al. (1984) beef bioaccumulation factors were used for antimony in the absence of soil invertebrate data. Beyer (1990) was used as the primary source for PAH bioaccumulation factors (Table 8-2). The average of all BAFs reported in Beyer (1990) was used to estimate total PAH BAF values for soil invertebrates. Note that BAFs presented in Table 8-2 are in units of wet organism, based on an assumption of 84 percent moisture in earthworms (EPA 1993).

8.2.1.2 Plant Bioaccumulation

Oak Ridge National Laboratory has derived similar regression equations for the prediction of metals in plants (Bechtel 1998). The equations presented in Table 7 of Bechtel (1998) have been used for the prediction of plant dry weight concentrations for copper, lead, and zinc (Table 8-3). The dry plant concentration has been converted to wet weight assuming that plant percent moisture is 75 percent (EPA 1993). The regression equations are as follows:

$$\ln [\text{plant}] = B_0 + B_1 (\ln[\text{soil}])$$

where:

B_0 and B_1 are contaminant specific parameters

$\ln[\text{soil}]$ is the natural log of the COPC concentration in soil

$\ln[\text{plant}]$ is the natural log of the calculated plant concentration

The plant concentrations (dry weight) were converted to wet weight by multiplying by the fraction dry of plants (assumed to be 25 percent based on 75 percent moisture value). The results of the regression calculations are presented in Table 8-7.

The two other primary sources for derivation of plant BAFs were Travis and Arms (1988) for organic chemicals and Baes et al. (1984) for metals not found in Bechtel (1998). Travis and Arms (1988) derived an equation that correlated plant concentrations with soil concentrations relative to the log K_{ow} of the analyte:

$$\log B_v = 1.588 - (0.578 \times \log K_{ow})$$

where:

$\log B_v$ = \log_{10} of the plant bioaccumulation factor

$\log K_{ow}$ = \log_{10} of the analyte K_{ow} .

As with soil invertebrates, the average PAH BAF has been used to estimate the bioaccumulation of total PAH.

Baes et al. (1984) derived an equation relating the concentration of metals in the vegetative portion of a plant with the soil concentration:

$$C_{\text{plant}} = C_{\text{soil}} \times B_v$$

where:

C_{plant} = concentration of analyte in plant (dry weight)

C_{soil} = concentration of analyte in soil

B_v = element transfer coefficient for vegetative portions of the plant from Baes et al. (1984)

The resultant plant concentrations for aluminum, antimony, chromium, and vanadium have been converted to a wet weight concentration assuming that plants are 75 percent water (EPA 1993).

8.2.1.3 Small Mammal Bioaccumulation

Few data are available for the assessment of bioaccumulation into small mammals (Table 8-4). Sample et al. (1998b) investigated models for the prediction of the bioaccumulation of metals into small mammals, and reported regression equations similar to those derived for soil invertebrates. The regression equations are as follows:

$$\ln [\text{small mammals}] = B_0 + B_1 (\ln[\text{soil}])$$

where:

B_0 and B_1 are contaminant specific parameters

$\ln[\text{soil}]$ is the natural log of the COPC concentration in soil

$\ln[\text{small mammals}]$ is the natural log of the calculated small mammal concentration

The small mammal concentrations (dry weight) were converted to wet weight by multiplying by the fraction dry of small mammals (assumed to be 32 percent based on 68 percent moisture value). The results of the regression calculations for chromium, copper, lead, and zinc are presented in Table 8-8.

The mean uptake factor (UF) from Appendix C in Sample et al. (1998a) was used for aluminum and vanadium. Antimony was not investigated by Sample et al. (1998a); therefore, Baes et al. (1984) beef transfer coefficient factors were used for antimony. The equation used was:

$$BAF_{\text{antimony}} = \text{Ingestion to beef transfer coefficient } (F_r) \times \text{Fraction dry}$$

where:

$F_r = 0.001$ for antimony

Fraction dry = 0.32

Beyer (1990) was used as the primary source for PAH bioaccumulation factors (Table 8-4). The average of all BAFs reported in Beyer (1990) was used to estimate total PAH BAF values for

small mammals. Note that BAFs presented in Table 8-4 are in units of wet organism, based on an assumption of 68 percent moisture in small mammals (EPA 1993).

8.2.2 Dose Equation

Dietary exposures for ROC have been estimated as body-weight-normalized daily doses for comparison to a body-weight-normalized daily dose toxicity reference value (TRV). Separate doses are presented for soil and food contributions, and these are summed to produce the total dose for each ROC.

$$Dose_{total} = Dose_{food} + Dose_{soil}$$

where:

- $Dose_{total}$ = Total daily dose of COPC received by receptor; mg COPC/kg-body wt/day
- $Dose_{food}$ = Daily dose of COPC received by receptor; mg COPC/kg-body wt/day from food items
- $Dose_{soil}$ = Daily dose of COPC received by receptor; mg COPC/kg-body wt/day from incidentally ingested soil

The total dose from food is given by:

$$Dose_{food} = AUF \times F_f \sum_{i=1}^n F_i \times C_{soil} \times BAF_i$$

where:

- AUF = Area usage factor (fraction of habitat range represented by site) for receptor (Table 8-1)
- F_f = Total daily feeding rate in kg wet food/kg-body weight of ROC/day (Table 8-1)
- F_i = Fraction of i^{th} prey item in diet (unitless) (Table 8-1)
- C_{soil} = Concentration of COPC in soil (mg/kg)
- BAF_i = Bioaccumulation factor for i^{th} organism (mg/kg wet organism/mg/kg dry soil)
(Tables 8-2 through 8-4)

The total dose from incidental soil is given by:

$$Dose_{soil} = F_s \times F_x \times AUF \times C_s$$

where:

- F_S = Total daily incidental soil feeding rate in kg soil wet/ kg-body weight of ROC/day (Table 8-1)
 F_X = Fraction dry of soil sample (unitless)
AUF = Area usage factor (fraction of habitat range represented by site) for receptor (Table 8-1)
 C_S = Concentration of COPC in soil; mg chemical/kg soil (dry basis)

8.3 EXPOSURE CONCENTRATIONS

As noted earlier, aluminum, antimony, chromium, copper, lead, vanadium, zinc, and Total PAH have been identified as COPC in the Tier I, Step 1 and 2 ERA. Analytical results for these analytes in 10 surface soil samples (0.5 ft bgs) collected in October 1999 (SB1 – SB9, including one duplicate for SB6) were used in this Step 3 ERA and food web. Results for SB6 were averaged and used as the representative concentration for that sample. Total PAH concentrations were determined by summing the individual PAHs for each individual sample as long as the specific PAH was measured in at least one of the site samples. If the individual PAH was not detected in an individual sample, one-half the detection limit was used as a surrogate concentration. PAHs that are never detected in site samples were not included in the Total PAH calculation.

Exposure point concentrations in surface soil were estimated as the 95%UCLM values for exposure to ecological receptors with very small AUFs relative to the size of PI-1 (meadow vole). For receptors with larger AUFs (robin, kestrel, shrew, cottontail, and red fox) the arithmetic mean was used as the EPC.

The first step in estimation of EPC was to determine whether medium-specific environmental data for a COPC are normally or log-normally distributed. This was accomplished with the Shapiro-Wilks W-test for distribution (Gilbert 1987). Results of this statistical test, shown in Table 8-5, showed that copper, zinc, and total PAH were normally distributed, while none of the other COPC fit either normal or log normal distributions.

For copper, zinc, and total PAH the following steps were undertaken to calculate the 95%UCLM:

- (1) Calculate the arithmetic mean of the data.
- (2) Calculate standard deviation of the data.

- (3) Determine the one-tailed t-statistic (Gilbert 1987).
- (4) Calculate 95%UCLM using the equation given below:

$$95\%UCLM = (\bar{x} + t s / \sqrt{n})$$

where:

95%UCLM	=	95th percentile upper confidence limit on the mean
\bar{x}	=	Mean of the untransformed data
s	=	Standard deviation of the untransformed data
t	=	Student-t statistic
n	=	Number of samples in the data set

For the remaining COPC where the data fit neither normal or log normal distributions, the 95%UCLM was determined using a standard bootstrap estimation. The bootstrap approach (Efron 1981) provides a convenient way to estimate the standard error of a sample statistic, $\hat{\theta}$, without making any assumptions of how the original data are distributed. The following is a brief description of how the standard bootstrap procedure is used to estimate the upper confidence limit (UCL) of a sample statistic (Singh et al. 1997).

Step 1. From the original sample $X_n = (X_1, X_2, \dots, X_n)$; where the deviates X_i are independently and identically distributed, draw a sample of n observations with replacement such that each observation has the same probability of being drawn ($= \frac{1}{n}$).

The new data set is called the bootstrap sample, and is typically denoted as

$$X_n^* = (X_1^*, X_2^*, \dots, X_n^*).$$

Step 2. Compute the sample statistic, $\hat{\theta}^*$, of interest (in this case the sample mean \bar{X}) from X_n^* .

Step 3. The procedures in Steps 1 and 2 are repeated N times (e.g., 2000) generating N bootstrap estimates of the sample statistic. The general bootstrap estimate is the arithmetic mean of

the N estimates, $\bar{\theta}_B = \frac{1}{N} \sum_{i=1}^N \hat{\theta}_i^*$. The bootstrapped standard error of $\hat{\theta}$, denoted by $\hat{\sigma}_B$, is given by

$$\hat{\sigma}_B = \sqrt{\frac{1}{N-1} \sum_{i=1}^N (\hat{\theta}_i - \bar{\theta}_B)^2}.$$

Step 4. Finally, the $(1-p)100$ percent confidence limits of $\hat{\theta}$ are given by

$$\hat{\theta} \pm z_p \hat{\sigma}_B.$$

Where z_p is the p^{th} quantile of the standard normal distribution.

Resultant 95%UCLMs and arithmetic means for all COPCs are shown in Table 8-5.

As discussed in Sections 8.2.1.1 – 8.2.1.3, concentrations of copper, lead, and zinc in the food items, soil invertebrates, plants, and small mammals, and chromium in small mammals were estimated using empirically derived equations relating soil concentrations to concentrations found in the food items. The results are shown in Tables 8-6 through 8-8 for soil invertebrates, plants, and small mammals, respectively, along with predicted food concentrations.

8.4 TOXICITY ASSESSMENT

EPA (1997) guidance specifies that a screening ecotoxicity value should be “equivalent to a documented or best conservatively estimated chronic No Observed Adverse Effect Level (NOAEL).” Since there is wide variation in the literature on NOAELs and since conservative estimates have been used to obtain both screening values and exposures, the HQ will also be calculated for conservatively estimated Lowest Observed Adverse Effect Levels (LOAELs) to provide some frame of reference for the results. Wildlife TRVs from Sample et al. (1996), shown in Tables 8-9 and 8-10 for NOAEL and LOAEL values of the COPCs, have been used for the Step 3 toxicity assessment.

8.5 RISK CHARACTERIZATION

Exposure point concentrations for surface soil, invertebrate, vegetation, and small mammals are presented in Table 8-11. Area use factors calculations used in the risk characterization are in Table 8-12. Risks based on the modified food-web for Site PI-1 are summarized in Table 8-13. Calculations of dose and hazard quotients for each ROC are shown in Appendix K. Based on this assessment, NOAEL HQ were below 1.0 for all receptor/COPC combinations with the exception of shrew/vanadium (NOAEL HQ = 1.41), shrew/aluminum (HQ = 113.64), vole/aluminum (HQ = 21.34), rabbit/aluminum (HQ = 1.00), robin/aluminum (HQ = 5.53), robin/chromium (NOAEL HQ = 1.73), robin/lead (NOAEL HQ = 3.04), and robin/zinc (NOAEL HQ = 3.57). The majority of LOAEL HQ did not exceed 1, indicating that the receptor/COPC pairs did not exceed those concentrations that represent the lowest observed adverse effect concentration. The red fox and kestrel are not at risk from any of these COPC.

The HQs for the shrew at PI-1 for vanadium is 1.41 and this suggests that a very small potential exists for an environmental effect on the shrew. The HQs for the robin range from 1.73 – 5.53, suggesting that a very small potential exists for an environmental effect on the robin. Using a conservative BAF of one for Total PAH still results in acceptable risk to all receptors. Aluminum risk may be due to high concentrations in the environment due to the local geology and this is further discussed in the uncertainty section.

8.6 UNCERTAINTY ANALYSIS

The inclusion of aluminum as a COPC represents some uncertainty in the risk analysis. Aluminum is the most common metallic element comprising 8 percent of the earth's crust (Bodek et al. 1988). The natural abundance of aluminum in soil in the project area is substantiated by the bedrock mineralogy and background concentrations of aluminum in soil. As noted in Chapter 3, the bedrock of the area predominantly consists of gneissic granite and granodiorite. These rock types are composed of plagioclase and potassium feldspars with more than 10 percent quartz. Minor minerals include hornblende, pyroxene, biotite, muscovite, and garnet (Colorado School of Mines 1955, Commonwealth of Virginia 1993). The chemical formulae of these minerals, listed below, demonstrate the abundance of aluminum that can be found in the local bedrock and soil:

K-felspar	KAlSi_3O_8
Plagioclase feldspar (albite)	$\text{NaAlSi}_3\text{O}_8$
Hornblende	$(\text{Ca}, \text{Na})_{2-3}(\text{Mg}, \text{Fe}, \text{Al})_5\text{Si}_6(\text{Si}, \text{Al})_2\text{O}_{22}(\text{OH})_2$
Pyroxene (augite)	$(\text{Ca}, \text{Na})(\text{Mg}, \text{Fe}, \text{Al})(\text{Si}, \text{Al})_2\text{O}_6$
Biotite	$\text{K}(\text{Mg}, \text{Fe})_2(\text{AlSi}_3\text{O}_{10})(\text{OH})_2$
Muscovite	$\text{KAl}_2(\text{AlSi}_3\text{O}_{10})(\text{OH})_2$
Garnet (almandine)	$\text{Fe}_3\text{Al}_2\text{Si}_3\text{O}_{12}$

Developed on these types of acidic igneous rocks in the climate of the project area, residual soil contains the clay minerals kaolinite predominantly and vermiculite commonly. These clay minerals are hydrous silicates that contain oxides of aluminum, magnesium, and iron. The predominant cations include aluminum in kaolinite, and aluminum, magnesium, and iron in vermiculite (Dragun 1998).

According to Shacklette and Boerngen (1984), the aluminum concentration in soil in the region is in the range of 5 to 10 percent, or 50,000 to 100,000 mg/kg. Background aluminum concentrations in soil at Fort Pickett, as discussed in Chapter 1, ranges from 2,230 to 25,100 mg/kg in surface soil, and is up to 58,000 mg/kg in total soil (WESTON 1999a). The higher concentration in total soil as compared to surface soil is expected as the aluminum tends to be leached from the upper soil horizon (Dragun 1998). With such high background levels of aluminum found in the project area, inclusion of aluminum as a COPC may reflect actual geological concentrations versus actual site contamination.

In addition, the EPA has been examining the toxicity of this common element in conjunction with toxicity and has concluded (Eco-SSL 2000, pp. 8-1):

“EPA recognizes that due to the ubiquitous nature of aluminum, the natural variability of aluminum soil concentrations and the availability of conservative soil screening benchmarks (Efroymson et al. 1997), aluminum is often identified as a contaminant of potential concern (COPC) for ecological risk assessments. The commonly used soil screening benchmarks (Efroymson et al. 1997) are based on laboratory toxicity testing using aluminum solution amendments to test soils. Comparisons of total aluminum soil concentrations to solution based screening values are deemed by EPA to be inappropriate.”

The Agency goes on further to state (EcoSSL 2000, pp. 8-2):

“Potential ecological risk associated with aluminum in soils is identified based on the measured soil pH. **Aluminum is identified as a COPC only for those soils with a soil pH less than 5.5.** [Original emphasis] The technical basis for this procedure is that the soluble and toxic forms of aluminum are only present in soil under soil pH values less than 5.5.”

Soil pH at Site PI-1 has been reported at 7.4; consequently, aluminum’s inclusion as a COPC represents significant uncertainty.

8.7 CONCLUSIONS

The small magnitude of no observed adverse effect level exceedances suggest that risks to ecological receptors, including the robin and shrew, from exposure to COPC at PI-1 are negligible, and that no action is necessary for the site.

TABLE 8-1 EXPOSURE FACTORS FOR SITE PI-1 TERRESTRIAL ECOLOGICAL RECEPTORS OF CONCERN

Exposure Factor	Receptor					
	American Robin	American Kestrel	Meadow Vole	Short-Tailed Shrew	Eastern Cottontail	Red Fox
Body Weight (kg)	0.0773 Mean for adult, both sexes, all seasons, PA location (EPA 1993)	0.119 Average value from CA, UT studies (EPA 1993)	.0355 Adult male average, all year, IN location (EPA 1993)	0.0174 Mean value for adult female in summer from PA study (EPA 1993)	1.134 Mean for adult male, MD, WV populations (EPA 1993)	3.94 Mean value for adult female, Iowa (EPA 1993)
Food Ingestion Rate (g wet/g-bw/day)	1.52 Mean value in the range for both sexes, free-living in KS location (EPA 1993)	0.31 Highest value for adult male in summer, OH location (EPA 1993)	0.30 Mean value for adults, Russia (EPA 1993)	0.62 Value for adults, both sexes, WI (EPA 1993)	0.085 Calculated using equation for herbivores in EPA (1993)	0.069 Adult nonbreeding, North Dakota captive (EPA 1993)
Incidental Soil Ingestion Rate (g wet/g-bw/day)	0.15 Based on 10.4% soil in diet of American woodcock (EPA 1993)	0.0047 No data available; assumed low rate of 1.5% soil in diet based on food items and capture mode.	0.0072 Based on 2.4% soil in diet of meadow vole (EPA 1993)	0.062 No data available; a relatively high rate [10% soil in diet] was assumed because of the preponderance of earthworms and soil invertebrates in diet.	0.0017 Used 2% soil in diet of woodchuck (Beyer et al. 1994)	0.00193 Based on 2.8 percent soil in diet of red fox (EPA 1993)

Exposure Factor	Receptor					
	American Robin	American Kestrel	Meadow Vole	Short-Tailed Shrew	Eastern Cottontail	Red Fox
Territory (ha)	0.15 Based on foraging home range, adults feeding nestlings (EPA 1993)	131 Based on mean for Michigan woodlots and fields. (EPA 1993)	0.019 Based on adult male in Virginia old field (EPA 1993)	0.15 Mid range for juveniles and adults, both sexes of New York old field (EPA 1993)	3.0 Mid range of Pennsylvania home range (EPA 1993)	700 Based on Minnesota adult female (EPA 1993)
Dietary Fractions	62 % vegetation 38 % invertebrates Based on adults Eastern US, averaged over a year. (EPA 1993)	50 % invertebrates 50 % small mammals Based on pine oak woodlands intake in spring (EPA 1993)	100 % vegetation Based on IL, bluegrass, % volume stomach contents in summer (EPA 1993)	87 % invertebrates 13 % vegetation Based on % volume in stomach contents in NY/ Nova Scotia study, June-October (EPA 1993)	100 % vegetation Based on MD/CT studies in forest (EPA 1993)	75 % small mammals 12 % invertebrates 13 % vegetation Based on mid range in IL farmland woods (EPA 1993)

TABLE 8-2 ECOLOGICAL INVERTEBRATE BIOACCUMULATION FACTORS

<i>Chemical</i>	<i>Bioaccumulation Factor (wet organism/dry soil)⁽¹⁾</i>	<i>Source</i>	<i>Comments</i>
PAHs			
Benzo(b)fluoranthene	0.051	Beyer (1990)	See notes below for calculation
Chrysene	0.028	Beyer (1990)	See notes below for calculation
Fluoranthene	0.013	Beyer (1990)	See notes below for calculation
Naphthalene	0.015	Beyer (1990)	See notes below for calculation. Anthracene values used as surrogate.
2-Methylnaphthalene	0.015	Beyer (1990)	See notes below for calculation. Anthracene values used as surrogate.
Phenanthrene	0.019	Beyer (1990)	See notes below for calculation
Pyrene	0.015	Beyer (1990)	See notes below for calculation
Total PAH	0.022	Beyer (1990)	Mean value of PAHs used for BAF for Total PAH
Metals			
Antimony	0.00016	Baes et al. 1984	Adapted from beef bioaccumulation, no receptor specific data available.
Chromium	0.176	Sample et al. 1998a	Mean UF from Table 11 in Sample et al 1998a *0.16
Copper	Regression Equation	Sample et al. 1998a	Regression equation (see Table 6)
Lead	Regression Equation	Sample et al. 1998a	Regression equation (see Table 6)
Vanadium	0.006	Sample et al. 1998a	Mean UF from Table C.1. in Sample et al 1998a *0.16
Zinc	Regression Equation	Sample et al. 1998a	Regression equation (see Table 6)

(1) Invertebrate assumed to be 84 % water (EPA 1993)

Beyer (1990)

BAF calculated as Mean Tissue Concentration/Mean Soil Concentration * 0.16 (fraction dry) from Table 25 of Beyer (1990).

Baes et al. (1984)

BAF for antimony calculated as:

Ingestion to beef transfer coefficient F_r * 0.16 (fraction dry)

F_r value = 0.001 found in Figure 2.25 in Baes et al. (1984).

Sample et al. (1998a)

See regression equations in Table 3-6 of this document or Mean UF from Table 11 or Table C.1 in Sample et al. 1998a * 0.16 (fraction dry).

TABLE 8-3 ECOLOGICAL PLANT BIOACCUMULATION FACTORS

<i>Chemical</i>	<i>Bioaccumulation Factor (wet plant/dry soil)⁽¹⁾</i>	<i>Source</i>	<i>Comments</i>
PAHs			
Benzo[b]fluoranthene	0.0025	Travis and Arms (1988)	Regression equation (see explanation below)
Chrysene	0.0049	Travis and Arms (1988)	Regression equation (see explanation below)
Dibenz[a,h]anthracene	0.0013	Travis and Arms (1988)	Regression equation (see explanation below)
Fluoranthene	0.0106	Travis and Arms (1988)	Regression equation (see explanation below)
Naphthalene	0.1106	Travis and Arms (1988)	Regression equation (see explanation below)
2-Methylnaphthalene	0.1106	Travis and Arms (1988)	Regression equation (see explanation below)
Phenanthrene	0.0227	Travis and Arms (1988)	Regression equation (see explanation below)
Pyrene	0.0108	Travis and Arms (1988)	Regression equation (see explanation below)
Total PAH	0.0343	Travis and Arms (1988)	Mean of all PAH
Metals			
Antimony	0.05	Baes et al. (1984)	B _v value
Chromium	0.0019	Baes et al. (1984)	B _v value
Copper	Regression Equation	Bechtel (1998)	Simple regression equation (see text)
Lead	Regression Equation	Bechtel (1998)	Simple regression equation (see text)
Vanadium	0.0014	Baes et al. (1984)	B _v value
Zinc	Regression Equation	Bechtel (1998)	Simple regression equation (see text)

(1) Plant percent moisture assumed to be 75 % (EPA 1993)

Travis and Arms (1988)

BAF(B_v) calculated as $\log B_v = 1.588 - 0.578 \log K_{ow}$, where K_{ow} is contaminant specific.

B_v then multiplied by 0.25 (fraction dry)

Baes et al. (1984)

BAF calculated as: $B_v \cdot 0.25$ (fraction dry)

B_v values found in Figure 2.1 of Baes et al. (1984).

Bechtel 1998

See regression equations in Table 3- 7 of this document.

TABLE 8-4 ECOLOGICAL SMALL MAMMAL BIOACCUMULATION FACTORS

Chemical	Bioaccumulation Factor (wet organism/dry soil) ⁽¹⁾	Source	Comments
PAHs			
Benzo[b]fluoranthene	0.102	Beyer (1990)	See notes below for calculation
Chrysene	0.056	Beyer (1990)	See notes below for calculation
Fluoranthene	0.025	Beyer (1990)	See notes below for calculation
Naphthalene	0.029	Beyer (1990)	See notes below for calculation. Anthracene values used as surrogate.
2-Methylnaphthalene	0.029	Beyer (1990)	See notes below for calculation. Anthracene values used as surrogate.
Phenanthrene	0.039	Beyer (1990)	See notes below for calculation
Pyrene	0.029	Beyer (1990)	See notes below for calculation
Total PAH	0.044	Beyer (1990)	Mean value of PAHs used for BAF for Total PAH
Metals			
Antimony	0.00034	Baas et al. (1984)	See notes below for calculation
Chromium	Regression Equation	ample et al. (1998b)	Regression equation (see Table 8)
Copper	Regression Equation	ample et al. (1998b)	Regression equation (see Table 8)
Lead	Regression Equation	ample et al. (1998b)	Regression equation (see Table 8)
Vanadium	0.004	ample et al. (1998b)	Mean uptake factor from Table C.1. in Sample et al.
Zinc	Regression Equation	ample et al. (1998b)	Regression equation (see Table 8)

(1) Small mammal assumed to be 68 % water (EPA 1993)

Beyer (1990)

BAF calculated as Mean Tissue Concentration/Mean Soil Concentration * 0.32 (fraction dry) from Table 25 of Beyer (1990).

Baas et al. (1984)

BAF for antimony calculated as:

Ingestion to beef transfer coefficient F_r * 0.32 (fraction dry)

F_r value = 0.001 found in Figure 2.25 in Baas et al. (1984).

Sample et al. (1998b)

See regression equations in Table 3- 8 of this document or Table C.1. in reference for vanadium (mean uptake factor * 0.32 (fraction dry))

TABLE 8-5 EXPOSURE CONCENTRATIONS FOR STEP 3 ERA

Analyte	Units	N	Normal Distribution		Log Normal Distribution		Distribution ⁽²⁾	Arithmetic Mean	Median	Min	Max	95% UCLM ⁽³⁾
			Shapiro-Wilk Statistic (W_x)	p_x ⁽¹⁾	Shapiro-Wilk Statistic (W_y)	p_y ⁽¹⁾						
ALUMINUM	mg/kg	9	0.5322	<0.001	0.6082	<0.001	Neither	4.52E+03	4.29E+03	1.84E+03	9.41E+03	5.80E+03
ANTIMONY	mg/kg	9	0.5704	<0.001	0.4698	<0.001	Neither	4.68E-01	4.40E-01	1.20E-01	8.25E-01	5.94E-01
CHROMIUM	mg/kg	9	0.5322	<0.001	0.5414	<0.001	Neither	7.47E+00	7.10E+00	4.60E+00	1.00E+01	8.49E+00
COPPER	mg/kg	9	0.9569	0.766	0.6247	<0.001	Normal	9.98E+00	5.90E+00	1.40E+00	3.31E+01	1.62E+01
LEAD	mg/kg	9	0.5557	<0.001	0.6582	<0.001	Neither	6.85E+01	8.38E+01	1.15E+01	1.18E+02	9.22E+01
VANADIUM	mg/kg	9	0.6075	<0.001	0.5637	<0.001	Neither	1.07E+01	9.20E+00	5.40E+00	1.82E+01	1.27E+01
ZINC	mg/kg	9	0.9604	0.802	0.5597	<0.001	Normal	6.70E+01	5.59E+01	4.10E+00	1.81E+02	1.03E+02
TOTAL PAHs (ND=1/2)	ug/kg	9	0.8687	0.119	0.3898	<0.001	Normal	4.13E+02	2.68E+02	5.20E+01	1.88E+03	7.67E+02

N = Total number of samples analyzed

W_x = Shapiro-Wilk statistic for un-transformed data (X)

W_y = Shapiro-Wilk statistic for the log-transformed data ($Y = \ln X$)

⁽¹⁾ The probability p that the data were sampled from the hypothesized distribution, if $p < 0.05$, the data do not fit the specific distribution.

⁽²⁾ Data are assumed to be drawn from a normal distribution, unless $p_x < 0.05$ and $p_y > 0.05$. If both p_x and p_y are < 0.05 , then neither distribution was assumed.

⁽³⁾ One-tailed 95% UCLM for the prescribed distribution.

If the data fit neither a normal or lognormal distribution, then the UCLM was determined by bootstrap with 2000 iterations.

ND = Not Detected

TABLE 8-6 SOIL INVERTEBRATE BIOACCUMULATION CALCULATION

95% UCLM Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [earthworm]	Earthworm (mg/kg dw)	Fraction Dry	Earthworm (mg/kg ww)
Copper	16.2	2.785	1.675	0.264	2.410	11.1	0.16	1.781867
Lead	92.2	4.524	-0.218	0.807	3.433	31.0	0.16	4.954292
Zinc	103	4.634	4.449	0.328	5.969	391	0.16	62.59026

Mean Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [earthworm]	Earthworm (mg/kg dw)	Fraction Dry	Earthworm (mg/kg ww)
Copper	9.98	2.300	1.675	0.264	2.282	9.8	0.16	1.567955
Lead	68.5	4.227	-0.218	0.807	3.193	24.4	0.16	3.898038
Zinc	67	4.204	4.449	0.328	5.828	340	0.16	54.35615

Assumptions: Invertebrates are 84% water (U.S. EPA 1993)
Regression model from Sample et al. 1998a
Regression model: $\ln(\text{earthworm}) = B_0 + B_1(\ln(\text{soil}))$
Earthworm concentration (ww) = Earthworm conc (dw) * Fraction dry

TABLE 8-7 PLANT BIOACCUMULATION CALCULATIONS

95% UCLM Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [plant]	Plant (mg/kg dw)	Fraction Dry	Plant (mg/kg ww)
Copper	16.2	2.785	0.669	0.394	1.766	5.9	0.25	1.46
Lead	92.2	4.523	-1.328	0.561	1.209	3.35	0.25	0.84
Zinc	103	4.634	1.575	0.555	4.147	63.3	0.25	15.8

Mean Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [plant]	Plant (mg/kg dw)	Fraction Dry	Plant (mg/kg ww)
Copper	9.98	2.300	0.669	0.394	1.575	4.83	0.25	1.2
Lead	68.5	4.226	-1.328	0.561	1.043	2.84	0.25	0.71
Zinc	67	4.204	1.575	0.555	3.908	49.8	0.25	12.5

Assumptions: Plant are 75% water (U.S. EPA 1993)

Regression model from Bechtel 1998

Regression model: $\ln(\text{plant}) = B_0 + B_1(\ln(\text{soil}))$

Plant concentration (ww) = Plant conc (dw) * Fraction dry

TABLE 8-8 SMALL MAMMAL BIOACCUMULATION CALCULATIONS FOR COPPER, CHROMIUM, LEAD, AND ZINC

95% UCLM Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [mammal]	Mammal (mg/kg dw)	Fraction Dry	Mammal (mg/kg ww)
Copper	16.2	2.785	1.4592	0.2681	2.205	9.1	0.32	2.9
Chromium	8.49	2.139	-1.459	0.7338	0.1096	1.12	0.32	0.357
Lead	92.2	4.524	0.0761	0.4422	2.0766	7.98	0.32	2.55
Zinc	103	4.634	4.4713	0.0738	4.813	123	0.32	39.4

Mean Exposure Point Concentrations

Analyte	Soil (mg/kg)	Ln[soil]	B ₀	B ₁	Ln [mammal]	Mammal (mg/kg dw)	Fraction Dry	Mammal (mg/kg ww)
Copper	9.98	2.300	1.4592	0.2681	2.075	8.0	0.32	2.55
Chromium	7.47	2.011	-1.459	0.7338	0.0157	1.02	0.32	0.325
Lead	68.5	4.227	0.0761	0.4422	1.945	7.0	0.32	2.24
Zinc	67	4.204	4.4713	0.0738	4.781	119	0.32	38.2

Assumptions: Small Mammals are 68% water (U.S. EPA 1993)

Regression model from Sample et al. 1998b

Regression model: $\ln(\text{small mammal}) = B_0 + B_1(\ln(\text{soil}))$

Small mammal concentration (ww) = Small mammal conc (dw) * Fraction dry

TABLE 8-9 LIST OF NOAEL TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING^(a)
 (mg/kg-bw/day)

COPC	Robin	Kestrel	Shrew	Vole	Rabbit	Red Fox	Comments
INORGANICS							
Aluminum	109.7	109.7	2.295	1.754	0.767	0.551	NOAEL for robin and kestrel based on ringed dove exposed to aluminum sulfate. NOAEL for shrew, vole, rabbit, and red fox based on mouse exposed to aluminum chloride.
Antimony	NA	NA	0.149	0.114	0.05	0.036	NOAEL for vole, shrew, fox, and rabbit based on mouse exposed to antimony potassium tartrate.
Chromium	1	1	7.21	5.51	2.41	1.73	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL for Cr+6. NOAEL for robin and kestrel based on duck NOAEL for Cr+3.
Copper	47	47	33.4	25.5	11.2	8	NOAEL for vole shrew, fox, and rabbit based on mink NOAEL for copper sulfate. NOAEL for robin and kestrel based on day-old chicks for copper oxide.
Lead	3.85	3.85	17.58	13.44	5.88	4.22	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on kestrel NOAEL.
Vanadium	11.4	11.4	0.428	0.327	0.143	0.103	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on mallard NOAEL.
Zinc	14.5	14.5	351.7	268.7	117.6	84.5	NOAEL for vole, shrew, fox, and rabbit based on rat NOAEL. NOAEL for robin and kestrel based on white leghorn hen NOAEL.
PAHs							
Total PAH	NA	NA	1.19	0.91	0.4	0.29	NOAEL based on benzo(a)pyrene for mouse.

(a) NOAEL TRVs from Sample et al. (1996). Values for the Red-tailed hawk used for the American kestrel.
 NA = No NOAEL Available

TABLE 8-10 LIST OF LOAEL TOXICITY REFERENCE VALUES (TRVs) FOR USE IN FOOD-WEB MODELING ^(a)
 (mg/kg-bw/day)

COPC	Robin	Kestrel	Shrew	Vole	Rabbit	Red Fox	Comments
INORGANICS							
Aluminum	1097	1097	22.952	17.538	7.674	5.515	LOAEL for robin and kestrel based on ringed dove NOAEL times 10. LOAEL for shrew, vole, rabbit, and red fox based on mouse LOAEL.
Antimony	NA	NA	1.487	1.136	0.497	0.357	LOAEL for vole, shrew, fox, and rabbit based on mouse LOAEL exposed to antimony potassium tartrate.
Chromium	5	5	28.88	22.07	9.66	6.94	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL for Cr+6. LOAEL for robin and kestrel based on duck LOAEL for Cr+3.
Copper	61.7	61.7	44	33.6	14.7	10.6	LOAEL for vole, shrew, fox, and rabbit based on mink LOAEL. LOAEL for robin and kestrel based on 1 day old chicks.
Lead	38.5	38.5	174.83	134.35	58.79	42.25	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on kestrel NOAEL times 10.
Vanadium	114	114	4.285	3.274	1.433	1.03	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on mallard NOAEL times 10.
Zinc	131	131	703.3	537.4	235.2	169	LOAEL for vole, shrew, fox, and rabbit based on rat LOAEL. LOAEL for robin and kestrel based on white leghorn hen.
PAHs							
Total PAH	NA	NA	11.89	9.09	3.98	2.86	Mouse LOAEL TRV for benzo(a)pyrene used as surrogate.

(a) LOAEL TRV data from Sample et al. (1996).

LOAEL values for the Red-tailed hawk used for the American kestrel.

NA = No LOAEL Available

TABLE 8-11 EXPOSURE POINT CONCENTRATIONS FOR STEP 3 REFINED FOOD WEB FOR SITE PI-1

Mean Concentrations

Chemical	Surface Soil Concentrations (mg/kg) dw	Fraction Dry	Surface Soil Concentrations (mg/kg) ww ⁽¹⁾	Water Conc. (mg/L)	Vegetation Conc. (mg/kg) ww	Invertebrate Conc. (mg/kg) ww	Mammal Conc. (mg/kg) ww
Aluminum	4.52E+03	8.55E-01	3.87E+03	0	4.52E+00	3.84E+01	5.28E+01
Antimony	4.68E-01	8.55E-01	4.00E-01	0	2.34E-02	7.49E-05	1.59E-04
Chromium	7.47E+00	8.55E-01	6.39E+00	0	1.40E-02	1.32E+00	2.55E+00
Copper	9.98E+00	8.55E-01	8.54E+00	0	1.21E+00	1.57E+00	3.25E-01
Lead	6.85E+01	8.55E-01	5.86E+01	0	7.10E-01	3.90E+00	2.24E+00
Vanadium	1.07E+01	8.55E-01	9.13E+00	0	1.47E-02	6.41E-02	4.27E-02
Zinc	6.70E+01	8.55E-01	5.73E+01	0	1.25E+01	5.44E+01	3.82E+01
Total PAH	4.13E-01	8.55E-01	3.53E-01	0	1.42E-02	9.09E-03	1.82E-02

95% UCLM

Chemical	Surface Soil Conc. (mg/kg) dw	Fraction Dry	Surface Soil Concentrations (mg/kg) ww ⁽¹⁾	Water Conc. (mg/L)	Vegetation Conc. (mg/kg) ww	Invertebrate Conc. (mg/kg) ww	Mammal Conc. (mg/kg) ww
Aluminum	5.80E+03	8.55E-01	4.96E+03	0	5.80E+00	4.92E+01	6.77E+01
Antimony	5.94E-01	8.55E-01	5.08E-01	0	2.97E-02	9.50E-05	2.02E-04
Chromium	8.49E+00	8.55E-01	7.26E+00	0	1.59E-02	1.49E+00	2.90E+00
Copper	1.62E+01	8.55E-01	1.39E+01	0	1.46E+00	1.78E+00	3.57E-01
Lead	9.22E+01	8.55E-01	7.88E+01	0	8.38E-01	4.95E+00	2.55E+00
Vanadium	1.27E+01	8.55E-01	1.09E+01	0	1.75E-02	7.62E-02	5.08E-02
Zinc	1.03E+02	8.55E-01	8.81E+01	0	1.58E+01	6.26E+01	3.94E+01
Total PAH	7.67E-01	8.55E-01	6.56E-01	0	2.63E-02	1.69E-02	3.37E-02

1 . Soil concentration (ww) calculated by multiplying soil concentration (dw) by fraction dry [assumed to be 0.855 (samples 14.5% moisture on average)].

TABLE 8-12 AREA USE FACTORS FOR STEP 3 REFINED FOOD WEB FOR SITE PI-1

Fort Pickett PI-1 = 0.33 hectare

Assumption: If the Area Use Factor is > 1, a value of 1 will be used.

Organism	Site Area (ha)	Home Range (ha)	Area Use Factor	Adjusted Area Use Factor (value of 0-1)
Short-tailed shrew	0.33	0.15	2.2	1
Meadow vole	0.33	0.019	17.3684211	1
Eastern cottontail rabbit	0.33	3	0.11	0.11
Red fox	0.33	700	0.00047143	0.000471429
Robin	0.33	0.15	2.2	1
Kestrel	0.33	131	0.00251908	0.002519084

Area Use Factor Calculations

Area Use Factor = Site Area/Home Range

TABLE 8-13 HAZARD QUOTIENT SUMMARY FOR STEP 3 REFINED FOOD WEB FOR SITE PI-1

TERRESTRIAL SPECIES
REFINED FOODWEB
FORT PICKETT, SITE PI-1

Ecological Contaminants of Concern	Shrew		Vole		Rabbit		Red Fox		Robin		Kestrel	
	NOAEL	HQ _n	NOAEL	HQ _n	NOAEL	HQ _n	NOAEL	HQ _n	NOAEL	HQ _n	NOAEL	HQ _n
Aluminum	113.64	11.36	21.34	2.13	1.00	0.10	0.01	0.00	5.53	0.55	0.00	0.00
Antimony	0.18	0.02	0.11	0.01	0.01	0.00	0.00	0.00	NA	NA	NA	NA
Chromium	0.15	0.04	0.01	0.00	0.00	0.00	0.00	0.00	1.73	0.35	0.00	0.00
Copper	0.04	0.03	0.02	0.02	0.00	0.00	0.00	0.00	0.07	0.05	0.00	0.00
Lead	0.33	0.03	0.06	0.01	0.00	0.00	0.00	0.00	3.04	0.30	0.00	0.00
Vanadium	1.41	0.14	0.26	0.03	0.01	0.00	0.00	0.00	0.12	0.01	0.00	0.00
Zinc	0.10	0.05	0.02	0.01	0.00	0.00	0.00	0.00	3.57	0.39	0.00	0.00
Total PAH	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	NA	NA	NA	NA

NA - Not Available

HQ_n- Hazard Quotient based on the NOAEL

HQ_i- Hazard Quotient based on the LOAEL

9. FEASIBILITY STUDY

The results of the RI field investigation and risk assessments were used to evaluate the need for remedial action at PI-1. This RI/FS was conducted in accordance with guidance developed for the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA). CERCLA was implemented through the National Contingency Plan (NCP) for Oil and Hazardous Substances, as amended in 1990.

As discussed in Section 6 concerning the human health risk assessment, the only case where non-carcinogenic risks were above the EPA HI threshold of 1.0 was for child residents. The sole driver was manganese, which had a cumulative HI of 1.4 across soil and groundwater pathways. The target specific HI's for soil exposure did not exceed 1.0; therefore, potential non-carcinogenic effects from exposure to soil at the site are of no concern. Based on Fort Pickett soil background data, manganese concentrations detected at the site in soil are consistent with background manganese concentrations. Therefore, no evidence exists to indicate a site-specific release of manganese in soil that could be considered responsible for manganese levels in groundwater. Rather, the occurrence of manganese in groundwater at levels that result in non-cancer risk greater than 1.0 is more likely to be representative of regional conditions unrelated to PI-1 (see Section 10.2).

No cancer risks for any receptor exceeded the U.S. EPA's target risk range of 10^{-6} to 10^{-4} . Ingestion of total soil and ingestion of groundwater for the resident (adult and child) and ingestion of surface soil for the adolescent trespasser were the only exposure scenarios for which potential cancer risks exceeded the de minimus 10^{-6} cancer risk. Arsenic and dioxin in soil, and aldrin, alpha-HCH, and heptachlor in groundwater were the primary risk drivers. Site arsenic concentrations were well within the background data range for Fort Pickett. All other exposure scenarios fell below this 10^{-6} level.

For ecological risks based on the modified food-web (Section 8), NOAEL HQ were below 1.0 for all receptor/COPC combinations with the exception of shrew/vanadium (NOAEL HQ = 1.41), robin/chromium (NOAEL HQ = 1.73), robin/lead (NOAEL HQ = 3.04), and robin/zinc (NOAEL HQ = 3.57). These exceptions do not necessarily indicate that an effect will occur, but only that a lower threshold of toxicity may have been exceeded. No LOAEL HQ exceeded 1, which indicated that no receptor/COPC pairs exceeded those concentrations that represented the lowest observed adverse effect concentration. The NOAEL HQs for the robin ranged from 1.73

– 3.57, which suggests that a very small potential exists for an environmental effect on the robin. However, the small magnitude HQ at no observed adverse effect level suggests that risks to ecological receptors, including the robin and shrew, from exposure to COPCs are negligible and that no further action is necessary for the site.

Based on these findings, the existing constituent concentrations in soil and groundwater at the site do not pose a significant risk to human health and the environment. Therefore, no remedial alternatives were evaluated.

10. SUMMARY AND CONCLUSIONS

10.1 SUMMARY

10.1.1 Site History

PI-1 (Former Maintenance Area) encompassed about one acre and is located at the north end of Fort Pickett, adjacent to Business Route 460. PI-1 was formerly used as a vehicle maintenance area and service station. Vehicles, including buses, were frequently parked onsite for maintenance. ERI (1997) identified this site as a BRAC parcel in the aerial photographic survey. Photographs from 1949 and 1951 showed two buildings, several vehicles, and equipment at a probable maintenance area. Areas with stains and disturbed ground were observed in the southern portion of the site where vehicles, mechanical parts, and waste oil containers were possibly stored. The largest building present on the site was established in 1937 and was removed in 1964. A concrete slab with soil-filled vehicle maintenance pit remains. The smaller building was used as a bus stop and is not a suspect contamination source. A 2.5-ft diameter, former water supply well was located on the west side of the concrete pad. An underground septic tank may be located about 45 ft west of the pad, based on a metal detector survey.

10.1.2 Hydrogeology

Groundwater at the site occurs within the overburden on the west side of the site, but the water table is within gneissic granite bedrock on the east side. The overburden likely provides recharge to the deeper, fractured bedrock aquifer. The depth to top of bedrock is irregular being shallower on the eastern side of the site. Two downgradient wells were completed to depths of 24 and 26 ft within the overburden. Another two wells were proposed, one upgradient on the east and one on the north side of the site, but could not be completed due to auger refusal on bedrock. Auger refusal on bedrock occurred at depths ranging from 8 to 16 ft without encountering the saturated zone during several attempts to drill these wells. The water table was approximately 15 ft deep at the site. The groundwater gradient is 0.025 towards the northwest. With an approximate aquifer hydraulic conductivity of 0.001 ft/min, the horizontal groundwater flow velocity in the saturated overburden is 0.2 ft per day or approximately 70 ft/year.

10.1.3 Field Investigation

The field investigation commenced on 25 October 1999, and it was completed on 12 November 1999. It included the completion of 8 soil borings and collection of soil samples for subsequent chemical analysis, surface soil sampling at 9 locations and subsequent analysis, installation of 2 monitoring wells and collection of groundwater samples for chemical analysis, and a topographic survey. Analyses included EPA TCL/TAL organic and inorganic compounds, PAHs, PCBs, TPH, and dioxin/furans.

10.1.4 COPC Occurrence and Migration

Analyte concentrations were compared to EPA RBCs, MCLs, and ecological screening criteria. Analytes considered as COPCs within site soil or groundwater included acetone, dibenzofuran, several PAHs and pesticides, dioxin/furans, and several TAL metals. Potential COPC transport pathways include air transport, surface runoff, and advective groundwater flow. Surface and subsurface soil sampling did not indicate the presence of concentrated contaminant sources at the site other than moderate concentrations of metals, which may reflect background levels. The presence of acetone in the surface and subsurface soil samples was likely due to laboratory or sampling effect, but was evaluated as a COPC. If present at the site, acetone would quickly volatilize from soil or dissolve into water.

Within surface soil, PAH COPCs predominantly were located at surface soil sample SB-7-0.5, which was within the former maintenance pit. This soil also contained dibenzofuran as a COPC. These elevated hydrocarbons are likely due to the nature of the fill material placed in the pit. Metal concentrations in this sample, however, were comparable to other soil at the site. One PAH, 2-methylnaphthalene was a COPC in SB-8-0.5 located next to the concrete pad. Dioxin was a COPC in SB6-0.5 also located adjacent to the pad. TPH was elevated in several surface soil samples. These COPCs and TPH may reflect residual contamination from former vehicle repair operations at the site. Potential migration of these compounds is primarily through erosion and surface transport of surface soil. These compounds strongly absorb to soil, and tend to persist in the environment undergoing slow biodegradation. The presence of pyrene in the downslope soil sample SB-9-0.5 may be due to site conditions.

Pesticide COPCs were detected at low levels in groundwater, but do not appear to be derived from a concentrated onsite source. It is known that these pesticides may have been used in past practices in the general area. These compounds strongly partition from water into particulate and

organic matter, therefore, they are not expected to migrate long distances with water in dissolved form. The organic matter content in the overburden of weathered bedrock is not high, being approximately one percent. The ultimate fate of the pesticides is limited sorption to soil, followed by slow biodegradation.

Considering the 14 COPC metals, in general metal concentrations were at similar levels in surface soil across the site and may reflect background concentrations. However, the lead level of up to 118 mg/kg and zinc up to 181 mg/kg may be due to site conditions. The high magnesium, not a COPC, in SB6-0.5 (455,000 mg/kg) may be due to residual contamination. The transport of metals via the surface water pathway may be a transport concern. The elevated lead and zinc in the downslope surface soil sample SB-9-0.5 may be due to site conditions.

The seven metal COPCs in subsurface soil, may reflect background conditions and not due to site impacts (see Section 10.2). Transport of metals in the subsurface is dependent on several physical and chemical processes. In general, the solubility of metals tends to increase with increasing acidity, and conversely, lower mobility under neutral and slightly alkaline conditions. The elevated manganese level in groundwater may be due to the relatively acidic groundwater (pH 4-5) mobilizing this metal from the soil. The potential for contaminants in groundwater to migrate offsite and the potential for risk can not be discounted.

10.1.5 Human Health Risk Assessment

The human health risk assessment was conducted to assess potential non-carcinogenic effects and cancer risks from potential current and future site exposure. The only case where non-carcinogenic risks were above the EPA cutoff of 1.0 was for child residents. The sole driver was manganese, which had a cumulative HI of 1.4 across soil and groundwater pathways. The target specific HI's for soil exposure did not exceed 1.0; therefore, potential non-carcinogenic effects from exposure to soil at the site are not of concern. Based on Fort Pickett soil background data, manganese detected at the site in soil is consistent with background. The site mean and maximum manganese concentrations were less than the respective mean and 95 UCLM of background for total soil. Therefore, there is no apparent site-specific release of manganese in soil that could be considered responsible for manganese levels in groundwater. Although background groundwater data are not available for the site, it is likely that the manganese concentrations in PI-1 groundwater are also consistent with background. Therefore, the occurrence of manganese in groundwater at levels that result in non-cancer risk greater than 1.0

are not attributable to PI-1, and are likely representative of regional conditions unrelated to PI-1 (see Section 10.2).

No cancer risks for any receptor exceeded the U.S. EPA's target risk range of 10^{-6} to 10^{-4} . Ingestion of total soil and ingestion of groundwater for the resident (adult and child) and ingestion of surface soil for the adolescent trespasser were the only exposure scenarios for which potential cancer risks above the de minimus 10^{-6} cancer risk. Arsenic and dioxin in soil, and aldrin, alpha-HCH, and heptachlor in groundwater were the primary risk drivers. Site arsenic concentrations were well within the background data range for Fort Pickett. The site mean and maximum arsenic concentrations were less than the respective mean and 95 UCLM of background for total soil. All other exposure scenarios fell below this 10^{-6} level.

A consideration of Virginia DEQ acceptable cancer risk policy to the results of this risk assessment finds that although some individual chemicals exceed the goal of 10^{-6} , none of the receptors has an unacceptable cumulative cancer risk (above 10^{-4}).

Therefore, as discussed above, the only COPCs with risks within U.S. EPA's risk targets of 10^{-4} to 10^{-6} for carcinogens and HI of 1.0 for non-carcinogens which are site-related are dioxins in soils and aldrin, alpha HCH, heptachlor, and heptachlor epoxide in groundwater. None of these exceed the risk targets. The only potential receptor populations for which these COPCs contribute risks within these risk targets are potential future resident children and adults, potential adolescent trespassers, and potential future commercial workers.

10.1.6 Ecological Risk Assessment

Several COPCs had HQ values above one including aluminum, antimony, chromium, copper, lead, vanadium, zinc, and Total PAH. It should be noted that the mean and maximum concentrations used in the food web modeling for aluminum, chromium, and vanadium were below the respective mean and 95% UCLM of the surface soil background values reported for Fort Pickett. This indicates the ROC could ingest soil at PI-1 and the risk would be below the risk from ingestion of background surface soil samples for these three metals. Based on this, the Steps 1 and 2 of the Tier 1 Screening Risk Assessment concluded COPCs antimony, copper, lead, zinc, and Total PAH remained as potential risks to several of the ROC.

As recommended in the Final Phase I RI for PI-1 (EA 2000) and various comments from U.S.EPA, Virginia DEQ (Appendix L) a Scientific Management Decision Point (SMDP) was

conducted. Based upon this decision, a refined, Step 3 of the Tier 2 ecological risk assessment was performed. Refinements include adding an area use factor for each receptor in the risk calculations, converting dry weight soil to wet weight concentrations as the exposure point concentrations, and addressing the bioavailability of the COPCs to the ROC. Based on this assessment, NOAEL HQ were below 1.0 for all receptor/COPC combinations with the exception of shrew/vanadium (NOAEL HQ = 1.41), shrew/aluminum (HQ = 113.64), vole/aluminum (HQ = 21.34), rabbit/aluminum (HQ = 1.00), robin/aluminum (HQ = 5.53), robin/chromium (NOAEL HQ = 1.73), robin/lead (NOAEL HQ = 3.04), and robin/zinc (NOAEL HQ = 3.57). The majority of LOAEL HQ did not exceed 1, indicating that the receptor/COPC pairs did not exceed those concentrations that represent the lowest observed adverse effect concentration. The red fox and kestrel are not at risk from any of these COPC.

The HQs for the shrew at PI-1 for vanadium is 1.41 and this suggests that a very small potential exists for an environmental effect on the shrew. The HQs for the robin range from 1.73 – 5.53, suggesting that a very small potential exists for an environmental effect on the robin. Using a conservative BAF of one for Total PAH still results in acceptable risk to all receptors. Aluminum risk may be due to high concentrations in the environment due to the local geology.

10.2 CONCLUSIONS

10.2.1 Background or Offsite Groundwater Quality

Background or upgradient groundwater quality data were not available in this RI for PI-1; however, groundwater analytical data are available from other BRAC sites at Fort Pickett. Considering the pesticide and manganese COPCs in groundwater at PI-1, Table 10-1 lists pesticide and manganese data from five monitoring wells at sites near the Blackstone Airport within Fort Pickett (WESTON 1999b). The four pesticide COPCs, aldrin, alpha-HCH, heptachlor, and heptachlor epoxide, were not detected in these offsite wells; however, 4,4'-DDD, alpha-chlordane, dieldrin, and endrin, were detected at similar concentrations. Dieldrin is the major metabolite of aldrin (Montgomery and Welkom 1990). Although dieldrin was a pesticide in use in the U.S. before 1971 (Briggs 1992), its presence may reflect a widespread historical use of aldrin at Fort Pickett, which may explain the presence of aldrin at PI-1. The pesticides aldrin, alpha-HCH, heptachlor, and heptachlor epoxide were available for use in the U.S. until the early 1980's. Likewise, the trace levels of other pesticides in groundwater at PI-1 and other areas of Fort Pickett is likely due to the historic use of pesticides on the post.

In addition, the elevated manganese levels in the offsite wells (Table 10-1), appears to be reflective of the natural and acidic groundwater quality in the region. The offsite total concentrations ranged from 171 to 894 µg/L. The manganese concentrations at PI-1 ranged from 46.9 to 495 µg/L, filtered and non-filtered. Therefore, manganese in groundwater at PI-1 is most likely a natural condition of the overburden-bedrock aquifer in the area.

10.2.2 Recommendations

Based on the findings of the human health and ecological risk assessments, the existing constituent concentrations in soil and groundwater at the site do not pose a significant risk to human health and the environment. Therefore, following the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 as amended by the Superfund Amendments and Reauthorization Act of 1986 procedures, no alternatives were chosen for a Feasibility Study and based on this information, No Action is recommended at Site PI-1.

TABLE 10-1 SUMMARY OF OFFSITE GROUNDWATER QUALITY DATA (a)

Analyte	Monitoring Well Location				
	BCT-4	EBS-308		BCT -11	
	MW-1	MW-1	SW-1	BCT11-MW-1	BCT11-MW-2
Detected Pesticides (ug/L)					
4,4'-DDD	0.085 J	0.13 J	BQL	0.16 J	0.15 J
Alpha-Chlordane	0.0071 J	BQL	BQL	BQL	BQL
Gamma-Chlordane	BQL	BQL	0.0064 J	BQL	BQL
Dieldrin	0.022 J	BQL	0.012 J	BQL	BQL
Endrin	0.064 J	0.14 J	0.029 J	BQL	BQL
Metals (ug/L)					
Iron	1610 K	129 K	167 K	22,000	
Iron (filtered)				208 K	197 K
Manganese	171 K	241 K	524 K	894 K	
Manganese (filtered)				633 K	659 K
Zinc	34.4 K	14.9 K	85.6 K	72.8 K	
Zinc (filtered)				29.4 K	26.1 K

(a) Background concentrations from WESTON (1999b)

BLANK CELL INDICATES NO ANALYSIS.

J = Estimated value

K = Analyte detected, but may be biased high.

BQL= Not detected below quantitative limit

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